SECTION 3

POTENTIAL SOURCES OF CONTAMINANTS
TO THE ENVIRONMENT AND MAN

3.0 <u>Potential Sources of Contaminants to the Environment and Man</u>

3.1 <u>Background Concentrations of Radionuclides and Trace Metals</u>

3.1.1 <u>Naturally Occurring Radionuclides</u>

Potassium-40 and radionuclides in the decay chains of uranium-238 and thorium-232 are the principal sources in the earth's crust of background radiation. Figures 3.1 and 3.2 show the uranium-238 and thorium-232 decay chains. Potassium-40 constitutes 0.0118 percent of naturally occurring potassium. Its half-life is 1.26×10^9 years and, upon decay, potassium-40 emits a 1.46 MeV gamma ray in 11 percent of its disintegrations. Table 3.1 lists the average concentration of and gamma-ray energy released by these radionuclides in one gram of rock. Table 3.2 lists the radionuclide content and dose equivalent rates from common rocks and soils. Potassium-40 and the thorium-232 decay chain each contribute about 40 percent of the dose rate at 3 feet above the ground while the uranium-238 decay chain contributes approximately 20 percent of the total dose rate.

Radon-222 occurs in the uranium-238 decay chain and has a half-life of 3.8 days. It is a noble gas and, upon decay, produces a series of short-lived, alpha-emitting daughters (see Fig.3.1). The average atmospheric radon concentration in the continental U.S. is 0.26 pCi/liter (Oa72). Under most conditions, the radon daughters contribute less than 10 percent (a few tenths of a μrem/hr) to the terrestrial external dose equivalent rate. However, inhaled radon daughters contribute a large fraction of the total dose equivalent rate to the respiratory tract: about 50 percent (90 mrem/yr) to the lung and nearly all of the dose (450 mrem/yr) to the segmental bronchioles (NCRP75).

Eighty-five percent of the surface area of the United States, and nearly all of its population, is underlain by rocks and soils of sedimentary origin. However, the correlation between the bedrock activity and the aboveground activity is not clear.

In most soils, the amount of water varies from 5 to 25 percent. The soil moisture attenuates gamma radiation from the soil. The potassium-40 dose equivalent rate can decrease by 30 percent when the soil water content increases from 0 to 30 percent (OA72). Moisture can retard the diffusion of radon into the atmosphere and reduce the exposure to airborne radon daughters. Since radon daughters account for 95 percent of the gamma-ray energy from the uranium-238 series, their accumulation in the ground increases
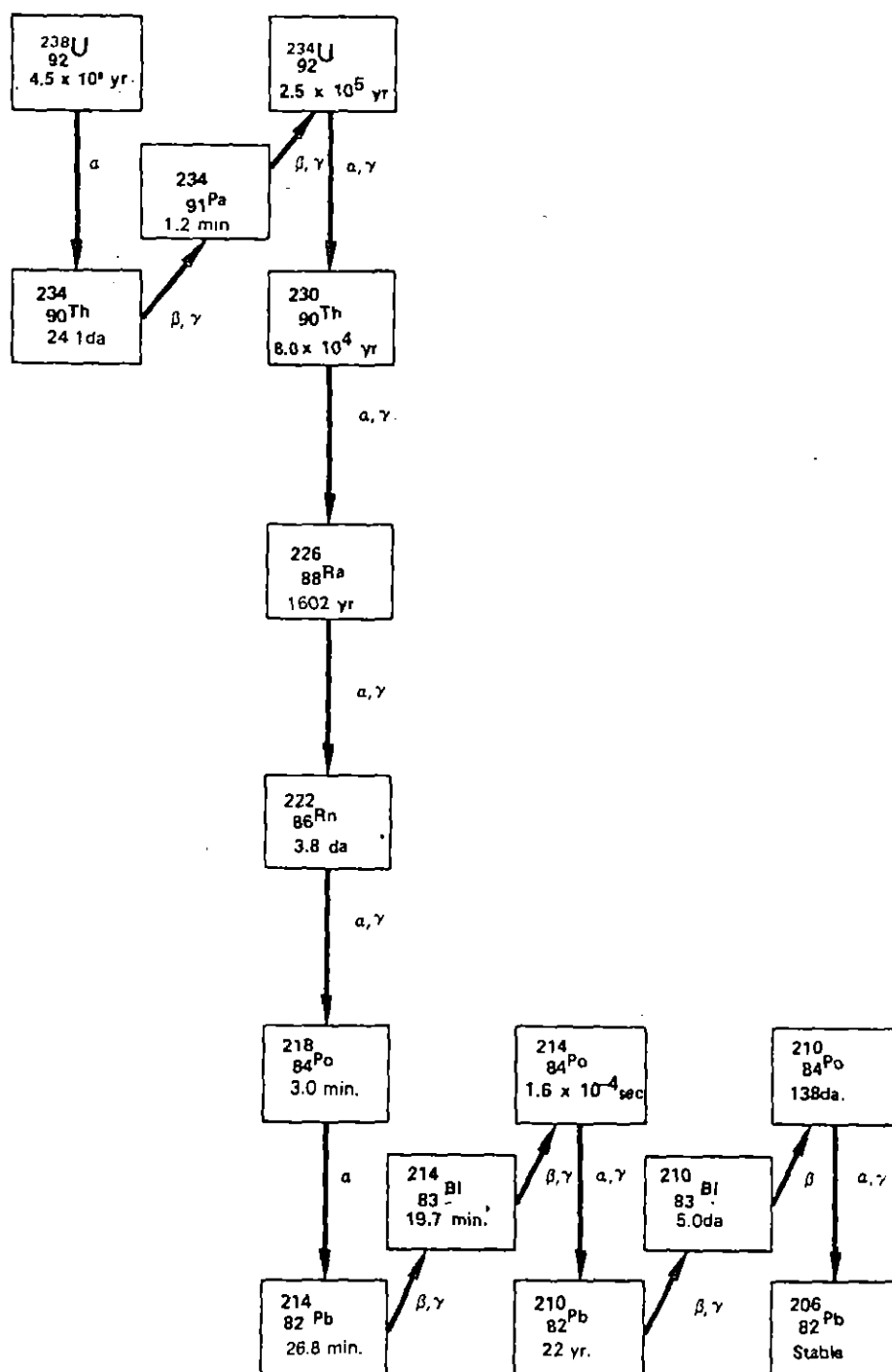
URANIUM — 238 DECAY SERIES

Figure 3.1 The uranium decay series showing the half lives and mode of decay.
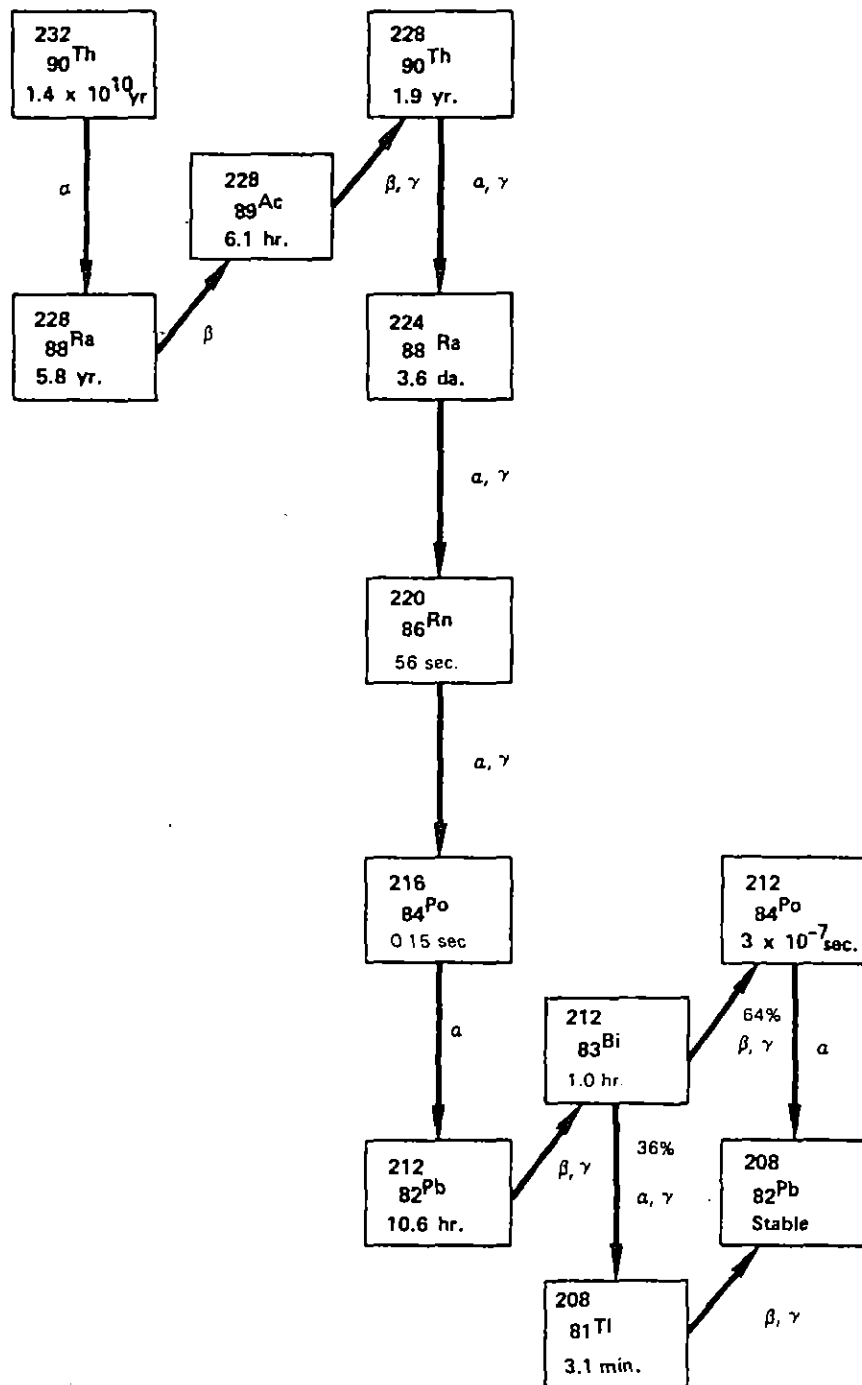
THORIUM - 232 DECAY SERIES

Figure 3.2 The thorium decay series showing the half lives and mode of decay.

Table 3.1 Gamma-ray energy released by one gram of rock

Isotope	Average Concentration, Percent	Energy, KeV/sec.
Uranium-238 (in equilibrium with decay products)	2.98×10^{-4}	68.2
Uranium-235 (in equilibrium with decay products)	0.02×10^{-4}	1.53
Thorium-232 (in equilibrium with decay products)	11.4×10^{-4}	87.8
Potassium-40	3.0	149
Other Elements	---	2.7

Source: Oa72.

Table 3.2 Radionuclide content and dose equivalent rates from common rocks and soil

Rock Type	Uranium		Thorium		Potassium-40		Total
	ppm	mrem/yr[a]	ppm	mrem/yr[a]	ppm	mrem/yr[a]	mrem/yr
Igneous[b] basic	0.9	5.2	2.7	7.3	1.2	14.7	27.2
Silicic (granite)	4.7	26.9	20.0	53.8	5.0	61.3	142.0
Sedimentary[b]							
Shale	3.7	21.2	12.0	32.3	3.2	39.2	92.7
Sandstone	0.45	2.6	1.7	4.6	1.1	13.5	20.7
Limestone	2.2	12.6	1.7	4.6	0.32	3.9	21.1
Upper crustal average[c]	2.8	16.0	10	26.9	2.4	29.4	72.3
U.S. surficial average[d]	1.8	10.3	9.0	24.2	1.8	21.8	56.3

[a]mrem/yr/ppm: uranium, 5.73; thorium, 2.69; potassium-40, 12.3 (Be68).

[b]Source: C166.

[c]Uranium and thorium averages (Ph64); potassium (He69).

[d]Source: Lo64.

the exposure from this series. Thus, soil moisture decreases the potassium-40 and thorium-232 dose equivalent rates and increases or leaves unchanged the uranium-238 series dose equivalent rate.

Snow cover also affects the terrestrial dose equivalent rate and the radon emanation rate. Gamma radiation attenuates exponentially as a function of the density and thickness of the snow cover (Oa72). However, the overall influence of snow on population exposure is negligible since, in most populated areas, there is relatively little snowfall that remains for long periods of time.

Table 3.3 shows the average dose equivalent rate due to radiation in some Western mining states (Oa72). Terrestrial radiation in the Western uranium mining states is higher than in the rest of the nation due to the greater concentration of the uranium-238 series.

Table 3.3 Average dose equivalent rates due to terrestrial
radiation in western mining states

State	Terrestrial Dose, mrem/yr
Arizona	45.6
Colorado	65.8
New Mexico	51.7
South Dakota	45.6
Texas	29.0
Utah	45.6
Wyoming	45.6

Concentrations of radionuclides measured in surface and groundwater samples collected on a proposed uranium project site are listed in Table 3.4 (NRC79a). The large variations among concentrations at different collection

sites are typical of surface water concentrations. (Concentrations in sea water are more uniform.) Hence, generalizations about background concentrations of radionuclides in fresh water systems are impractical. Extensive, site-specific studies over an extended period of time are necessary to obtain meaningful background concentrations for a site.

3.1.2 Stable Elements

Concentrations of metals occurring in the earth's crust generally range from several parts-per-billion (ppb) to a few parts-per-million (ppm). Measured concentrations vary widely from site to site and often in different samples taken from the same site. Table 3.5 lists the results of measurements for selected elements. It should be emphasized that these are general estimates of element composition of rocks in the United States and do not reflect large variations that occur within the different rock types.

Concentrations of metals measured in surface and groundwater samples collected from different locations on a proposed uranium project site are listed in Tables 3.6 and 3.7, respectively. There are large differences in the composition of surface and groundwaters. Table 3.8 shows the average concentrations of three trace metals that are sometimes associated with mine discharge water. These values, which were taken from the results of an extensive study (Tu69), approximate average concentrations in United States streams. Background concentrations at any specific site could be much different.

Table 3.8   Estimated average concentrations (ppb) of three
metals in U.S. streams

Element	Turekian's Results	Other results from Literature
Chromium	1.4	1.0
Molybdenum	1.8	1.0
Selenium	0.2	0.2

Source: Tu69.

Table 3.4 Radionuclide concentrations in surface and groundwater
in the vicinity of a proposed uranium project

Concentrations, pCi/ℓ

Radionuclide	Location 1	Location 2	Location 3	Location 4	Location 5	Location 6
			Surface Water			
U-238	9.8	4.0	2.5	1.8	1.6	---
Ra-226	0.4	0.5	<0.1	0.08	<0.1	---
Rn-222	145	108	42	<4	<4	---
Th-230	<0.1	0.2	0.3	<0.1	<0.1	---
Th-232	<0.1	<0.1	<0.1	<0.1	<0.1	---
			Groundwater			
U-238	2.0	4.5	2.3	3.2	3.8	1.2

Source: NRC79a.

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Table 3.7   Concentrations of selected elements in groundwater at six locations in the vicinity of a proposed uranium project

Element	Concentrations, mg/ℓ					
	Location 1	Location 2	Location 3	Location 4	Location 5	Location 6
Aluminum	<0.1	1	0.8	14.7	0.2	10
Antimony	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Arsenic	<0.01	<0.01	<0.01	<0.01	0.01	0.04
Barium	<0.1	0.5	0.5	0.2	0.3	0.7
Beryllium	0.001	<0.001	<0.001	0.003	0.001	<0.001
Cadmium	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cobalt	<0.01	<0.01	<0.01	0.01	<0.01	<0.01
Chromium	<0.001	0.003	0.003	0.004	0.001	0.003
Copper	0.003	0.003	0.006	0.027	<0.001	0.006
Lead	0.003	<0.001	<0.001	0.003	<0.001	0.002
Mercury	<0.0004	0.0012	0.0005	<0.001	<0.0004	0.0017
Molybdenum	0.001	0.002	0.002	0.001	0.001	0.003
Nickel	<0.01	<0.01	<0.01	0.02	<0.01	<0.01
Selenium	0.06	<0.01	<0.01	<0.01	0.13	<0.01
Vanadium	0.01	0.01	0.02	0.01	0.02	0.08
Zinc	4.4	0.07	0.21	0.07	0.01	0.02

Source: NRC79a.

3.2 Water-Related Aspects of Uranium Mining

3.2.1 Previous and Ongoing Hydrologic and Water Quality Studies Related to Uranium Mining

In the late 1950's and early 1960's, the U.S. Public Health Service conducted field studies to determine the water quality impacts of the uranium mining and milling industry. The studies emphasized uranium milling rather than mining. The Federal Water Pollution Control Administration conducted extensive stream surveys to assess the effects of uranium milling (but not mining) on the main stem and principal tributaries of the Colorado River. Subsequent stream survey work in Colorado by the Water Pollution Control Commission and the U.S. Geological Survey (Mo74, We74) mentioned a portion of the Uravan Mineral Belt and uranium mines therein, but the work did not emphasize uranium mines. Significant amounts of acidity and total trace metal concentrations were found in streams from 18 different mining areas. Dilution and chemical precipitation below mine drainages decreased concentration and increased the pH. Given enough time and distance, the streams recover naturally, but the accumulations of trace metals in the sediments increase. Field observations in 1971-72 of streams in most of Colorado indicated that approximately 724 km of streams in 25 different mining areas were adversely affected by mine drainage (We74).

Discussions of the impacts of uranium mining on water quality or quantity are incidental in numerous impact statements and environmental reports prepared by industry and (or) the U.S Nuclear Regulatory Commission as an integral part of licensing or relicensing uranium mills. Coverage on mining is usually minor as the principal focus is on milling impacts. The same is true for the recently prepared generic EIS on regulation of uranium milling (NRC79b).

Radiochemical assessment of surface and groundwater in uranium mining districts of New Mexico is done by self-monitoring programs associated with NPDES permits. Also, radiochemical assessment studies have been funded recently by the New Mexico Environmental Improvement Division and U.S Environmental Protection Agency. Self-monitoring, particularly in the pre-operational phase, characterizes mining and milling operations in all of the concerned States. These, together with results of surveys by State personnel, have resulted in extensive files of water quality data, flow measure-

ments, field observations of mine conditions, and exchanges between industry, regulatory agencies, and the public. Rarely are the data assembled and interpreted for dissemination outside a given agency. States experiencing rapid growth in uranium mining and milling are undoubtedly placing first priority on activities directly related to licensing, monitoring, and otherwise implementing regulations. Unfortunately, there is no concerted effort to prepare broad assessments of the cumulative impacts of mining and milling. Texas, New Mexico, and Wyoming are cases in point. Critical review and synthesis of these types of data can produce rather useful information. For example, the publication "Water Quality Impacts of Uranium Mining and Milling Activities in the Grants Mineral Belt, New Mexico" (EPA75) addresses the groundwater and surface water changes as the result of extensive uranium mining and milling production in a relatively confined area.

Some states have since initiated review of their data files, conducted field studies, and, in some cases, contracted study teams to investigate similar water quality changes. For example, a recent report by the Wyoming Department of Environmental Quality summarizes 16 years of aqueous radium and uranium data. The study reports that significant amounts of Ra-226 and uranium were present in surface water in the Shirley Basin as a result of inadequate mine water treatment (Ha78).

In Texas, surface water and groundwater monitoring conducted by industry, as well as by State and Federal agencies, reveal little or no change of chemical quality attributable to uranium mining and milling (Ge77, Ka76). This conclusion is based on 586 samples collected from 198 stations over a period of 39 years but primarily from 1961 to 1975. The State monitoring program by several agencies is continuing, but either summary reports are not issued or are two years overdue, depending on the agency. Not all of the findings exonerate the industry. One survey (It75) showed that none of the mine water from the 10 lakes that were sampled was suitable for human use. The lakes were also unsuitable for irrigation due to mineralization of the water by sulfate, chloride, and TDS. One of the 10 was suitable for stock watering.

The conditions or limits in the NPDES permits consider the quality of water being discharged, the quality of receiving water, and available, practical, treatment technology. Industry is required to monitor the discharges

on a periodic basis, usually daily, weekly, or monthly, and report results to EPA. The NPDES permits and related monitoring data help to estimate the quantity and quality of discharge allowed to enter the off-site environment.

Intensive studies of the influence of uranium mining on water quality and availability have not been conducted. Most investigations to date have been site-specific, of relatively short duration, and focused on the influence of surface discharge or subsurface seepage on water quality. Baseline studies from specific projects typically consist of quarterly or semi-annual sampling and are oriented toward milling instead of mining activities. The effects of dewatering on depleted water supplies or on water quality shifts in the aquifers of an area are rarely considered, and, then only on a mine-by-mine basis. Rarely are soil, stream sediment, and biologic samples collected in the preoperational period for radiologic analysis.

3.2.2   Mine Water Management

Figure 3.3 shows a scheme for considering the fate of water discharged from underground and open pit mines, including principal sources and sinks, most of which affect both water flow and quality. Broken lines in Fig. 3.3 indicate less important sequences with respect to water quality. For example, those mines that handle all water by on-site evaporation are likely to involve small volumes of water, and impacts on groundwater as a result of seepage are also likely to be small.

Mine drainage is surface water or groundwater flowing from a mine or an area affected by mining activities. Mine related point and nonpoint pollution sources can contaminate both surface water and groundwater throughout all phases of mining, that is, during mineral exploration, mine development, mineral extraction, processing, transport, and storage, and waste disposal. While mine-related point pollution sources usually include only milling and processing plant discharges and mine dewatering discharges, nonpoint sources can occur during any or all phases of mining. The chemical and physical characteristics and the mode of transfer of these nonpoint sources are variable and depend upon, among other things, the mineral being mined, its geo-

Figure 3.3. Disposition of drainage water from active surface and underground uranium mines

logic environment, the interrelations of all associated hydrologic systems (both surface water and groundwater), and the type of processing, transportation, storage, and waste disposal methods. Some mine-related nonpoint pollution sources are as follows (EPA77a):

1. suspended solids carried by immediate surface runoff
2. dissolved solids carried by immediate surface runoff
3. suspended and dissolved solids in proximate subsurface water seepage
4. dissolved solids in groundwater recharge
5. dissolved solids in groundwater discharge
6. uncontrolled contributions from mine-related point sources:
   a. high instantaneous concentrations of regulated pollutants in excess of effluent discharge guidelines, but falling within the NPDES instantaneous and daily average discharge limitations
   b. unregulated minor contaminants in point source discharges which are not specifically included under NPDES effluent limitations
   c. untreated mine dewatering discharges during or following major storm events (NPDES point source treatment systems may be bypassed during storm events of greater than a 10-year, 24-hour intensity)
7. reclaimed mine area and undisturbed area drainage diversion discharges
8. surface water and groundwater contamination and degradation induced by mine-related hydrologic disturbances and imbalances

Typically, waters affected by mine drainage are chemically altered by an increase in iron, sulfate, acidity (or alkalinity), hardness, TDS, and various metals, and are physically altered by an increase in suspended solids such as silt and sediment (Anon69,Hi68).

Many but not all uranium mines dewater at rates of 1 to perhaps 20 m^3/min. Typically, the water from the mine goes to settling ponds and then either to the mill or a nearby stream, dry wash, river, etc. Depending on the amount of mine water recycled in the mill and the amount of water pumped from the mines, there may or may not be any release to streams or arroyos. In at least one instance in the Grants Mineral Belt, mine water is totally recycled through the mine to enhance solubilization of uranium which is removed with ion exchange columns. Large evaporation ponds and some seepage losses help maintain a water balance and minimize releases to streams, arroyos, etc. Increasing competition for water in the western states is likely to induce maximum mine water reuse (in the mill), reinjection, or use

for potable supplies (Hi77) or power plant cooling.

When mill tailings ponds are used for final disposal of mine water, there is significant addition of chemical and radiochemical contaminants to the water in the course of milling. After treatment to reduce suspended solids, mine water may be recycled for use in the milling process or released to nearby streams, necessitating radium removal and reduction of suspended solids. Dissolved uranium in mine water, if present in concentrations exceeding about 3 mg/ℓ, is recovered by ion exchange columns. Settling ponds at the mines remove suspended solids. The water is then conveyed to receiving streams or to the mills for uranium recovery and (or) to satisfy mill feed water requirements. There are rather rigid requirements for release to surface water compared to groundwater.

A recent survey of 20 U.S. uranium mills (Ja79a) found large variation in the degree of water recycling. Where mine water is readily available, it probably is reused less than in water-short areas. More efficient water use by uranium mills possibly could increase the amount of (relatively) high quality mine water being discharged to the environment; lessen adverse impacts of mill tailings disposal by reducing the amount of liquid; and make mine water available as a source of potable water (after treatment) in water-short areas such as Churchrock and Gallup, New Mexico (Hi77). To date, water quality deterioration related to seepage and accidental release of tailings to surface streams has received the most study and has been the focus of regulatory programs. In the future, it is likely that water quantity issues will become increasingly important, particularly in areas where water supplies are already limited and where extensive dewatering necessarily accompanies mining.

Of 20 uranium mills surveyed, 6 reported part or all of the mill feed water came from mine drainage (Ja79a). In New Mexico, 19 of 30 mines surveyed by the State Environmental Improvement Division (J. Dudley, written communication) had off-site discharge to arroyos ranging up to 19 m^3 per minute. Those mines with no discharge utilized evaporation ponds or used the water for dust control. Most of the mines discharged to arroyos. In several instances, however, water was piped to a nearby mill at flow rates of 5 to 8 m^3 per minute. Relatively small quantities of mine water were used for sand backfill of mines, in-situ leaching of old workings, and irrigation of grasslands. In summary, New Mexico mines discharge 66 m^3 per minute off-site.

Of this, 12 m^3 per minute is routed to mills, and the balance is discharged directly off-site from the mine. Average discharge to streams and arroyos for the active underground mines in the Grants Mineral Belt on the whole was 1.8 m^3/min, whereas 12 mines in the Ambrosia Lake District averaged 1.7 m^3/min. In New Mexico, all mines discharging to an arroyo practice radium removal with approximately 90% efficiency. Uranium removal from mine water discharge occurs in all but two active mines. Future trends are likely to reflect increased discharge from the mine to the environment. Settling ponds, radium removal, or both will be used to meet discharge permit requirements.

In Wyoming, discharge from both surface and underground mines may be used as process water for uranium mills, discharged to surface streams, or used for irrigation. For example, at the North Morton underground mining operation, approximately 2 m^3 per minute of mine water discharge will be used to irrigate 800 hectares of alfalfa. At the South Morton surface mine operation, a like amount of discharge will become mill feed water.

A survey of all active U.S. uranium mills showed that 14 of 20 make no use of mine water (Ja79a). This may reflect mines where water simply is not encountered or the fact that mines and mills are not co-located. Most mills depend on deep wells, except in New Mexico where mine water is the main mill water supply. Table 3.9 summarizes water sources for U.S. uranium mills. Proposed NRC regulations on mill tailings disposal (44 Fed. Reg. 50012-59) purport to make long-term tailings isolation the primary consideration in mill siting. In areas subject to severe natural erosive or dispersive forces, this may mean that mills cannot be sited in the vicinity of mines. This may have effects on use of mine water for milling.

Table 3.9 Summary of feed water sources for active U.S. uranium mills

Water Source	No. of Mills
Rivers, Reservoirs	4
Wells	8
Springs	1
Unknown	1
Mine Water	3
Mine Water and Wells	3
	20

Source: Ja79a.

Although underground mining is now dominant in the Grants Mineral Belt, the greatest number of mines are small stripping operations that have long been inactive. This type of mining activity has apparently had little adverse impact on water resources. Few data are available on drainage assessment of large open pit mines such as the Jackpile-Paguate. The St. Anthony pit discharges about 0.076 m^3 per minute. Usually, the ore is above the water table. Any water present on the mine floor presumably is flood runoff or discharge from a nearby underground mine. Other strip mines in the Mineral Belt were not studied; hence, no conclusions were drawn (J. L. Kunkler, USGS, in preparation).

Mine dewatering is done either by pumping the mine pit/shaft directly or by drilling high capacity wells peripheral to the mine and pumping a sufficient volume of water to at least partially dewater the sediments. Because of the great volume of water that must be removed from an aquifer, the latter method is impractical for deep underground uranium mines. This is particularly true for the artesian aquifers of most of the deeper mines in the Grants Mineral Belt. More commonly this method is reserved for shaft sinking and open pit mines to depths of several hundred feet. Most underground mines are dewatered by pumping the water that collects in the mine itself. Borings ("longholes") made into the ore body for assay work and explosives facilitate drainage. There is considerable difference in the quality of water depending on the dewatering method used. Water removed from wells adjacent to the mine typically is representative of natural quality, but water removed from the mine can be high in radionuclides, stable elements, and suspended solids. In large part this is due to the disruptive nature of mining. However, more subtle, chemical processes of oxidation and bacterial action, aided by evaporation and free flow of air in the mine, are also operative.

The extent to which uranium exploration adversely impacts water resources is not well understood. Land surface disruption from drilling pads and access roads obviously affects erosion rates and results in mud pits and piles of contaminated cuttings on or near the land surface. Subsurface effects are less obvious. A potentially serious one is interaquifer connection via exploratory boreholes. In Wyoming, 6 million meters of exploratory drilling took place in 1979. Although State law requires mining companies to plug the holes after drilling, it is common practice to install only a surface plug and to rely on the drilling mud to effect a seal at depth.

Similar situations are likely in New Mexico and Texas. Shortages of funds and personnel to oversee proper completion and abandonment exist at the State level.

Hydraulic effects of water released from mines, whether from pumping or gravity flow, include increased surface discharge, recharge of shallow aquifers by infiltration, and decline of static water levels in formations intersected by the mine or related cone of depression. Of most concern are the effects relating to mine water discharge on downstream users and any influences, direct or indirect, of pumping/dewatering on water quality in the ore body and contiguous strata. In some locations, the Grants Mineral Belt, for example, the ore body is also a major regional artesian aquifer; hence, dewatering affects present water levels and will affect water levels at least to the year 2000, with complete recovery taking much longer. The extent and significance of uranium mine dewatering are as yet poorly documented. Recent studies have been made in New Mexico where dewatering is of concern because of the influence on regional groundwater availability for municipal use and in relation to return flows to the San Juan River (NRC79b; Ly79). In Wyoming, static water levels in wells on ranches adjacent to uranium mines owned by Exxon, Kerr-McGee, Rocky Mountain Energy Co., and other companies southwest of Douglas and between Pumpkin Buttes and Douglas have reportedly dropped 7 to 10 meters (Anon79). Water quality changes associated with dewatering generally are unknown and not specifically monitored regardless of the mining area location.

Water quality associated with dewatering is generally good, although suspended solids may be high, as expected. Discharge from dewatering wells will be low in suspended solids because of filtering by soil and rock aquifers. Overall water quality from dewatering wells, particularly for underground mines, is likely to be representative of ambient conditions in the ore body and, to a lesser extent, the adjacent formations that may also be dewatered.

Recent USGS work on groundwater in the San Juan Basin Region has indicated that mining expansion will have a significant impact on the water yield of the Morrison Formation (Ly79). In this study, although no water quality data are derived, the recharge and mine dewatering parameters that impact the expected drawdowns in the aquifer imply that a total of 7.03×10^8 m^3 of water will be produced by the 33 planned or announced mines by the year

2000. If the projected development of 72 mines occurs, dewatering would exceed 1.48×10^9 m^3. The model also estimates that flow in the San Juan River will decline very slightly (0.05 m^3/min). Similarly, flow in the Rio Grande Valley would be reduced by 0.85 m^3/min. The impacts will continue after mining and dewatering cease.

Table 3.10 summarizes New Mexico uranium mine discharge in relation to mine type, depth, and status (active or proposed). Note that projected mining is primarily underground and represents an average increase in mine depth of 275 percent and an increase in dewatering rate from 2.4 to 13.8 m^3 per minute. One would expect numerous water quality and quantity issues to arise if these projections materialize. For example, competition for water supply is likely to be widespread throughout the Upper Colorado River Basin, and uranium mines/mills are already relatively large water users. Dewatering and discharge require no water rights under State water laws in New Mexico, but the water is essentially wasted. Use of water in mills constitutes a beneficial use of water, and state water laws therefore require filing for water rights. Such filings may be denied upon protest from existing water users.

Inactive uranium mines and related wastes also influence water quality, particularly as a result of chemical and physical transport by surface water runoff. The main reasons why mine waste piles erode more quickly than undisturbed soils are lack of topsoil, steep angle of slopes, presence of toxic elements and buildup of salt in the near surface, and poor water retention characteristics. Usually, inactive surface and underground uranium mines are not a source of direct discharge of water, be it contaminated or of ambient quality, because of the low rainfall-high evaporation characteristics of the western uranium regions, static groundwater levels deep below the land surface in mining areas, and, in a few instances, recontouring of mined lands such that drainage is internal. Whether mines contaminate groundwater by groundwater leaching or by recharge contacting exposed oxidized ore bodies is poorly documented. Preliminary feasibility studies by the U.S. Geological Survey (Hi77) indicate generally good quality water from one inactive underground mine in the Churchrock area of New Mexico. It is possible that this water may be used as a municipal water supply for Gallup, New Mexico.

Table 3.10 Current and projected uranium mine discharges
in the Grants Mineral Belt, New Mexico

Mine Type	Number of Mines	Average Depth (m)	Average Discharge (m^3/min)
Active			
Underground	33	248	2.42
Open pit	3	48	0.045
Proposed			
Underground	46	681	13.8
Open pit	0	N/A	N/A

Source: Environmental Improvement Division, State of New Mexico.

For most uranium regions, the volume of discharge from inactive mines to surface water bodies, though poorly documented, is believed to be less significant than that from active mines. The degree to which inactive mines contribute contaminants, directly or indirectly, to adjacent water resources can only be qualitatively assessed. The significance of inactive mines is highly dependent on regional setting and mine type.

Inactive surface mines in Texas are, with rare exception, not a source of direct discharge to surface water. It is unknown if there is any adverse impact from standing water in the mine pits, the most recent of which have been final-contoured with an internal drainage plan. Various observers suspect that water quality deteriorates when overland flow crosses mine spoils associated with overburden piles (It75 and He79). Water in the mine pits is unsuitable for potable and stock use due to high stable element contents, but it is generally acceptable in terms of radioactivity. Water in Texas open pit mines is a combination of runoff and groundwater. Before release from a mine, water is put in retention ponds to reduce total suspended solids. Holding ponds are used for storing mine water, and discharge is not allowed unless such discharge does not adversely affect the receiving

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Environmental problems associated with alkaline and saline drainages are not well documented (Hi73).

Water quality impacts from uranium mining are a function of both quality and quantity of discharge. Underground and surface mines commonly require dewatering prior to or during the ore removal phase, although there is considerable variation from one area or mine to another. In New Mexico, large and medium sized mines are essentially dry, whereas mines in Texas and Wyoming require extensive dewatering. Regardless of location, underground mines rarely are dry and many require extensive dewatering. Considering the variety of water management measures, regional differences in contaminants and receiving waters, and geochemical characteristics of ore bodies, detailed discussion of the effects of mine drainage or mining, in general, on water quality must await further site - or area - specific study. It is questionable if sufficient data on mine drainage exist to assess effects on biota and the fate of contaminants in surface or sub-surface water bodies.

Limited data from Colorado, Texas, Wyoming, and New Mexico suggest adverse impacts on water quality from discharge of mine water. Effects of dewatering on deep groundwater quality are very poorly documented; hence, no conclusion as to relative significance is drawn. With respect to surface water resources, discharge of mine water and overland movement of water and suspended or dissolved contaminants may be significant. Because of the dearth of data on overland flow, emphasis herein is on contaminants discharged via mine drainage water. Substantial studies to evaluate sediment yield and quality from lands mined for uranium, particularly from areas of surface mining, have not been conducted, although recent work in Texas (He79) is a notable exception.

Elements such as uranium, radium, molybdenum, selenium, zinc, and vanadium may be enriched in point and nonpoint discharges from uranium mines. The dispersal, mobility, and uptake of such elements are directly relevant to the subject of this report. We reviewed selected literature and field data to at least qualitatively understand what processes and elements are most significant and to thereby strengthen some of the underlying source term assumptions in the transport and health effects modeling. Despite the annual chemical load introduced to ephemeral streams by both dissolved and suspended constituents in mine effluent and overland flow from mined lands, waste piles etc., a number of processes affect the concentrations in the ambient environ-

ment. These include dilution, suspended sediment transport, sorption and desorption, precipitation, ion exchange, biological assimilation or degradation, and complexation.

3.2.3.1.1 Dilution and Suspended Sediment Transport

In many uranium mining regions there is flooding or flash flooding. Such storms may well be the only runoff event for a year or more at a time. It is worthwhile then to consider some of the effects of such events on mobilization of contaminants associated with uranium mining wastes. Typically there is significant interdependence between the physical and chemical processes.

A principal physical process is dilution. This will reduce concentrations of pollutants released to surface waters, but is considered to play a relatively minor role over the short term for water percolating through the soil to sources of groundwater in arid or semiarid regions.

Transport of suspended sediments in floods is another dominant process. Suspended load is largely a result of physical, hydraulic processes, hence elements that are rapidly and thoroughly removed from solution as a result of solubility limits, precipitation with other ions, ion exchange, and sorption may well be transported in the suspended load. In metal mining areas of central Colorado, total and dissolved metal loads in streams are greater during high flow periods, apparently a result of flushing from mines and tailings piles and scouring of chemical precipitates from stream substrates (Mo74). Typically, total and dissolved loads decrease downstream, regardless of discharge. Increase in iron in the downstream direction reflects scouring of precipitate--an amorphous, hydrated ferric oxide. Dispersal occurs for quite a distance downstream.

3.2.3.1.2 Sorption and Desorption

Sorption can play an extremely important role in purifying waters, particularly if infiltration or percolation is involved. This is especially true when contaminant concentrations are too low to undergo precipitation reactions. Virtually every ionic species will be sorbed and removed to some extent except for chloride and, to a lesser extent, sulfate and nitrate. These seem to pass through soils and alluvium without significant sorption (Ru76). Sorption processes can be highly specific, depending on the type of

contaminant and the physical and chemical properties of both the solution and the porous medium.

There have been numerous laboratory studies on the sorption, leachability, and mobility of stable elements in various types of soil. A recent review of the literature on radionuclide interactions in soils specifically discusses radium, thorium, and uranium, which are especially pertinent to uranium mining (Am78). Distribution coefficients, the ratio of concentration in soil to that in water, ranged between 16 and 270 for uranium in various soil-river water systems, between 200 and 470 for radium, and on the order of 10^5 for thorium at pH6. These observations appear consistent with the generally accepted ideas that uranium is relatively mobile, thorium extremely immobile, and radium somewhere in between in natural water systems (NRC79b; Ku79; Ga77a).

Adsorption is believed to be important for cadmium, copper, lead, and nickel, insofar as these are transported in the suspended fraction, whereas manganese and zinc are primarily in the dissolved fraction. Adsorption of metals onto precipitated manganese oxides or hydroxides at elevated pH is probably insignificant in the case of most uranium mine discharges insofar as these are alkaline and, furthermore, discharge to or co-mingle with other streams that are alkaline.

Radium sorption and desorption tests done on uranium mill waste solids and river sediments collected from several locations in the Colorado Plateau (Sh64) and in Czechoslovakia (Ha68) showed that leaching is primarily controlled by the liquid-to-solid ratio, i.e., the volume of leaching liquid per unit weight of suspended solids. Natural leaching from mining and milling waste solids freely introduced to rivers in the past is one of the major factors in radium contamination of rivers (Ru58). Although settling ponds are now used to remove or at least reduce suspended solids from active mine discharges, the dissolved radium load sorbed on sediments presents a source term that may be somewhat analogous to river sediments contaminated by dissolved and suspended milling and mining wastes. The manner in which the radium is mobilized and the significance is poorly understood and bears further investigation. Apparently there is a "leap frog" transport mechanism involving combined chemical and physical weathering processes. There should be a marked downstream attenuation of both dissolved and sorbed/precipitated radium inventories insofar as sediment burial and dilution take place and leachability limits are reached, i.e., no more radium can be removed regard-

less of the duration, frequency, or intensity of agitation. Should the stream eventually discharge into a reservoir, it is unlikely that renewed leaching will take place. Shearer and Lee (Sh64) did not account for some factors that may be locally significant, such as bio-uptake along the stream/ river, use of water for irrigation, number of uranium facilities discharging, and other local factors.

Experiments were conducted in Japan (Ya73) to determine uranium adsorption and desorption using carbonate solutions and three soil types (alluvial, sandy, volcanic ash). Very high adsorption ratios and very low desorption ratios of uranium characterized the various soil types in contact with stream water and help explain the decrease in soluble uranium with flow distance from mines (Ma69). When wastewater flows into streams at the maximum permissible concentration (1.8 mg U/ℓ) recommended (ICRP64), Yamamoto et al. (Ya73) conclude that the uranium behaves as a uranyl carbonate complex anion and that essentially complete sorption readily occurs in the presence of (Japanese river) water which contains 15 to 39.9 mg/ℓ bicarbonate. Since this is similar to concentrations in surface waters of uranium regions in the western states, similar results are expected.

Sorption or desorption of heavy metals such as Co, Ni, Cu, and Zn in soils and fresh water sediments occurs in response to the aqueous concentration of metal, aqueous concentration of other metals, pH, and amount and strength of organic chelates and inorganic complex ion formers in solution (Je68). Other controls on the heavy metal concentrations in soil and fresh water include organic matter, clays, carbonates, and oxide and hydroxide precipitates.

To what degree solubility acts as a limit on stable element concentrations in natural waters is unclear. The crystallographic form or even the chemical composition of a precipitate are often unknown. Elements such as iron, aluminum, manganese, and titanium form insoluble hydroxides and are likely to exceed equilibrium solubility limits (An73). Hem (He60) partially disagrees, saying "it is not unreasonable to assume equilibirum for the iron species in water." Whether mine discharges or overland flow from mined areas are in equilibrium is unknown, but it is doubtful considering the underground or flash-flood origin of such waters. The non-equilibrium aspects of certain peak runoff events has been documented for major streams of the world (Durum and Haffty, 1963). Metals such as iron, aluminum, manganese, and titanium,

which readily form rather insoluble hydroxides as particulates or colloids, may be dissolved from suspended minerals during high flow conditions. Organics present in such flood waters may assist through formation of soluble complexes. Resulting metal concentrations may be higher than solubility and redox relationships alone would indicate.

3.2.3.1.3.   Precipitation

Probably one of the most significant processes affecting stable element solubility in natural water systems is adsorption on hydrous ferric and manganese oxides. Jenner (Je68) believes this is the principal control on the fixation of Co, Ni, Cu, and Zn (heavy metals) in soils and fresh water sediments. For example, ferric hydroxide adsorbs one to two orders of magnitude more SeO_3 per unit weight than clays, and 90 to 99 percent adsorption is possible at a pH of seven to eight typical of most western streams (Ho72). At neutral or slightly alkaline pH, both iron and manganese are poorly soluble in oxidizing systems and, in general, exhibit very similar chemical behavior, although manganese is slightly more soluble. Fixation of selenium in soils, particularly by iron oxide or as ferric selenite, renders it unavailable to agricultural and forage crops, although specific selenium-accumulating plants can remove the element and, upon decomposition, release it in water soluble forms, such as selenate and organic selenium compounds, available to other plants (Ro64). The behavior and mechanism of selenium adsorption (as selenium oxyanion) by hydrous ferric oxides is readily extended to the interpretation of other similarly bound minor elements (Ho72). Mobile selenium oxyanion in slightly alkaline waters might be carried to streams by surface runoff or in groundwater. Selenite selenium sorbed upon ferric hydroxide should be transported in surface waters at neutral or slightly acid pH. Other metals forming highly insoluble hydroxides in the pH range of 6 to 9 include copper (above pH 6.5), zinc (above pH 7.5), and nickel (above pH 9). Molybdenum is thought to hydrolyze to the bimolybdate ion under acid conditions and precipitate with iron and aluminum. Aerobic or oxidizing conditions in the vadose zone are favorable for the development of many of these oxides (Cu, Fe, Mn, Hg, Ni, Zn, Pb). Reducing conditions deep in saturated zones generally lead to increased mobility of these metals.

Reducing conditions that can exist in the presence of organic material (bituminous or lower ranking coals, anaerobic bacteria, fluidized humates)

can lead to precipitation reactions favorable for removing contaminants from mine waters. Reduction of uranium to the quadrivalent state and its fixation on clays would play the major role in protecting groundwater supplies from uranium if the appropriate reducing agents were present in soils (Ga77a; Ku79). Inorganic reducing agents could include ferrous iron and hydrogen sulfide produced by the action of an aerobic bacteria on sulfates. Natural reducing conditions can also, theoretically, cause the formation of such native elements as arsenic, copper, mercury, selenium, silver, and lead, which are all quite insoluble in their elemental form (Ru76). Hydrogen sulfide or other sulfides, if available, will serve to reduce the concentrations of such metals as arsenic, cadmium, copper, iron, lead, mercury, molybdenum, nickel, silver, and lead.

The metal-scavenging of hydrated iron oxide precipitate has been documented in a mined area of Colorado where relatively acid schists and gneisses give rise to acid runoff that dissolves large quantities of aluminum, magnesium, and zinc. Runoff from a nearby drainage basin underlain by basic rocks containing base and precious metal veins carries considerably less metal. However, manganese oxide precipitated with iron oxide contains large quantities of metals. Ferric hydroxide precipitates from aerated water solutions containing more than 0.01 ppm iron at pH values of 4.5 and above, aluminum hydroxide precipitates in the pH range of 5 to 7, and manganese hydroxide precipitates above pH8 (He60; Ch54). Considering the alkaline pH of most uranium mine discharges and overland flow from non-point sources such as mine waste piles, precipitation of iron and possibly manganese seems certain. The scavenging effect of iron hydroxide at neutral to alkaline pH is considerably less than that of manganese hydroxide precipitate.

The extensive studies of mine drainage in Colorado by Morgan and Wentz (1974) revealed the effects of solubility on stable element transport. In the downstream direction, dilution and neutralization of the acid mine drainage by bicarbonate caused dissolved metal to decrease due to dilution, chemical precipitation, and probably adsorption onto ferric hydroxide precipitate. The latter creates a coating on the stream substrate for a considerable distance during low flow periods. Subsequently, flood events scour and transport the precipitates. Manganese and zinc remain primarily in the dissolved phase for a considerable distance, whereas cadmium, copper, iron, lead, and nickel concentrate in the suspended fraction and, when turbulence

decreases, precipitate. The mobility sequence for the metals studied in Colorado generally follows the order Mn ≅ Zn > Cu > Cd > Fe > Ni > Pb. Ferric hydroxide precipitation and scavenging seems to be more important at neutral than at acidic pH's (Je68).

3.2.3.1.4 Biological Assimilation and Degradation

Biological uptake and the role it has on stable element concentrations in water is not predictively understood (An73). Plant uptake of stable elements and resulting phytotoxicity is not merely a function of how much is present in the soils or water. In the case of arsenic, the chemical form of arsenic appears more important than the total soil arsenic (Wo71). For example, water-soluble arsenic in soil created more phytotoxic effects than those with no detectable water-soluble arsenic. Soils high in reactive aluminum remained less phytotoxic, despite heavy applications of arsenic, than soils with low reactive aluminum. Selenium in soils can be present as elemental selenium, selenates, pyritic selenium, ferric selenites, and organic selenium compounds of unknown composition. Selenates and organic compounds are most available to plants, although slow hydrolysis of the other forms can occur such that they become available for plant uptake. The importance of water soluble selenium versus total selenium as the major factor affecting plant uptake has been demonstrated (La72; Gr67). Where sufficient selenium is present in plant-available form, all species will take it up in sufficient amounts to be harmful to animals (La72). Naturally occurring soils containing such available forms are geographically confined to semiarid regions or areas of impeded drainage. Such soils are not hazardous to humans and only locally are they a threat to animals.

Despite numerous examples of high selenium (up to 2.7 ppm) in surface water, particularly that associated with drainage from seleniferous soils in agricultural areas, Rosenfeld and Beath (Ro64) reported only a few cases of water-related selenosis in man or livestock. Water high in selenium is typically unpalatable to livestock and certainly to man. Lakin (La72) concluded that environmental contamination due to selenium is increasing, but hazardous concentrations are unlikely; mining and industrial wastes may cause local problems; and the effect of added selenium in waters in combination with other contaminants bears further study.

Uranium uptake by several species of native plants in the southeastern

Utah portion of the Colorado Plateau varied, sometimes strikingly, with the species, time of year, part of plant, availability of uranium in the soil, and chemical composition of the underlying rocks (Ca57). The type of rooting system and the soil moisture conditions also were influential. In some cases, there was no consistent relationship between the amount of uranium in the soil versus that in the plant ash. Plants are much less selective with respect to cadmium uptake, and it has been conclusively demonstrated that plants absorb cadmium from cadmium containing solutions and soils (Pa73; Fu73). Phytotoxic effects vary considerably with plant species. Cadmium and zinc sulfides tend to concentrate in the organic matter of soils. Upon oxidation to sulfate, plant availability increases along with solubility. Under alkaline conditions (pH8), cadmium is taken up rapidly by biota and by sediments. However, modeling of cadmium transport and its deposition in aquatic systems is very complex and encompasses many variables, most important of which are pH, carbonate content, chemical form, and competing ions.

3.2.3.1.5 Complexation

Published data on Gibbs free energies, enthalpies, and entropies of 42 dissolved uranium species and 30 uranium-bearing solid phases were recently reviewed (La78). Uranium in natural waters is usually complexed with carbonate, hydroxide, phosphate, fluoride, sulfate, and perhaps silicate. Such complexes greatly increase the solubility of uranium minerals and increase uranium mobility in groundwater and surface water. In waters with typical concentration of chloride, fluoride, phosphate, and sulfate, intermediate Eh's, neutral to alkaline pH's, and the presence of phosphate or carbonate, uranyl phosphate or carbonate complexes form and increase mineral solubility by several orders of magnitude. Sorption of the uranyl minerals carnotite, tyuyamunite, autunite, potassium autunite, and uranophane onto natural materials is greatest in the pH range of 5 to 8.5. Uranium content of small streams, in particular, can exhibit wide spatial and temporal variations due to pH and oxidation state of the water, concentrations of complex-forming species such as carbonate or sulfate, and presence of highly sorptive materials such as organic matter, certain metallic hydroxides, and clays (La78). Whereas sorption is probably a dominant control on stable element concentrations in low temperature aqueous conditions, there is insufficient information concerning specific sorbents to allow accurate prediction.

3.2.3.2 Results of Field Studies in Uranium Mining Areas

3.2.3.2.1 Colorado

Extensive studies of the effect of mine drainage on stream water quality and biota were done in central Colorado (Mo74). Although uranium was mined in 14 of the 25 areas studied, other metals were the principal products. Most of the ores were high in iron sulfides, and associated drainage was acidic. Also studied, but less intensely, was the Uravan district of western Colorado where the principal products are uranium and vanadium from Mesozoic sandstone. The Uravan Mineral Belt is different in terms of principal product and geologic features from other mining areas studied in Colorado. For these other areas, the drainage is acidic and heavily enriched in heavy metals and, therefore, somewhat atypical of most Colorado uranium mines in the Uravan area.

After a preliminary field survey of the temperature, specific conductivity, pH, stream-bottom conditions, and aquatic biota at 995 stream sites, 192 were chosen for detailed sampling and analysis during 1971-1972. The data indicate the contamination of approximately 711 kilometers of streams in 25 different areas, mostly in the Colorado Mineral Belt. The water quality effects in these areas arise from many varied causes, including active and inactive mine drainage, tailings pond seepage, drainage tunnels, and milling operations. The length of the streams affected is not absolute as it varies with the time of the year and flow conditions (Mo74).

The general findings indicate that Mn, Se, and SO_4 concentrations, and specific conductivity are poor indicators of mine drainage as natural sources can cause high values for these parameters even in undisturbed areas. Uranium mines make at least some contribution to problems of contaminated streams in central Colorado. In central Colorado, the exact impact of uranium mining on stream water quality is unknown but believed to be less important or significant in most areas as compared to impact from other mining, with the possible exception of the Boulder-Jamestown area (J. Goettl and D. Anderson, Colorado Game, Fish, and Parks Division and Water Pollution Control Commission, respectively, personal communication). Cadmium, As, and Pb exceed the U.S. Public Health Service toxicity limits, respectively, 12.5 percent, 1.4 percent, and 2.1 percent of the time. Mercury and Ag limits were never exceeded, and Cr was never detected. Iron and Mn standards were

frequently exceeded by large percentages, however, these limits are only based on aesthetics. Concerning the negative impacts of the various constituents, Cu and Zn (exceeding the limits 7.8 and 9.0 percent of the time, respectively) pose the greatest threat to resident aquatic life. Mining operations in the Uravan area are a relatively minor source of metals for the San Miguel River (Mo74). Potential problem areas are settling ponds and tailings piles associated with the mining operation. Although not a source of acid drainage, these sources did cause increased concentrations of copper, iron, manganese, nickel, vanadium, and zinc in the river. Only manganese exceeded the standard for drinking water, and no metal concentrations exceeded the biological criteria. Seepage (0.003 m^3/s, pH 6.8, 3300 mg/ℓ HCO_3) from a mine tailings area into Atkinson Creek, a tributary of the San Miguel River, observed in December 1972 caused no adverse impacts.

Because of its size, proximity to population, and effects on surface water quality, extensive surface water quality investigations to assess the impacts of mine water discharge from the Schwarzwalder mine have been made (EPA72). Grab samples of the mine effluent taken in 1972 revealed 15 mg/ℓ uranium and 80 pCi/ℓ radium-226. As of 1972, overflow and seepage from the settling ponds used to treat the mine effluent significantly degrade the radiochemical quality of nearby Ralston Creek. This was confirmed by both EPA and the State/Denver Water Boards monitoring program. With 20-fold dilution, Ralston Creek downstream of the mine contained 3 pCi/ℓ and 82 µg/ℓ dissolved radium-226 and uranium, respectively. With no dilution, as during July, concentrations were 81 pCi/ℓ and 20,300 µg/ℓ. Influx of contaminated stream water to nearby Long Lake raised dissolved radium-226 to 0.8 pCi/ℓ (4-fold increase over background) and uranium to 230 µg/ℓ (20 times background). From these data, conclusions were reached that the mine water caused a 5 percent increase in the radiation dose to consumers in a local water system (based on FRC and NCRP standards and daily consumption of 1.0 liter water). If the 4.5 mg/ℓ uranium limit proposed by ICRP was used, the estimated dose increases to nearly 40 percent of the dose limit for a population group. Since 1972, the effluent has been treated for radium-226 and uranium removal. Trace metals analysis of water samples collected from the creek and the water treatment plants revealed concentrations comparable to or greater than those in the effluent as of July 20, 1978. Concentrations

(μg/ℓ) were as follows:

	As	F*	Pb	Se	Zn
Mine Effluent	5	1	15	<2	18
Ralston Creek (avg)	5	1.3	32	<2	56
Water treatment plants (avg)	5	0.55	97	2	146

*mg/ℓ

3.2.3.2.2 Wyoming

We assessed the effects of mine drainage by literature review and a limited field study in the Spring of 1979. Results of the latter conclude this section of the report. The effects of mine dewatering, in situ leaching, and mill tailings seepage on surface water quality in the Shirley Basin were previously studied by the Wyoming Department of Environmental Quality (Ha78). Sixteen years of data on aqueous radium and uranium indicated significant amounts of radium-226 and uranium reached streams because of inadequate mine dewater treatment, mill tailings pond seepage, and improper operation of a precipitation treatment unit. Uranium concentrations in stream water increased 60-fold because of mine water discharge and possible tailings pond seepage. The effects of past loadings of uranium and radium on fish propagation or migration are not clear, although biologic uptake of uranium and radium has occurred. Phytoplankton, algae, and bottom fauna organisms also do not appear to have been adversely affected, but no studies have been conducted since 1962. Long-term effects of increased radioactivity levels are known and merit further study: "There exists real need for additional studies to determine the mechanisms involved in the dispersion and ultimate disposal of uranium loaded into the drainage basin...Only after additional studies have been completed, may we understand the total and long range impact that the company's activities have had on the aqueous environment" (Ha78). This latter finding related specifically to the current (1978) loading of uranium from treated mine discharge.

Previous studies by the State of Wyoming (Ha78) found that solution

mining by the Pathfinder Uranium Company noticeably affected ambient uranium concentrations in the study areas. A 1968 survey by the Department of Environmental Quality (Ha78) indicated relatively high loadings of soluble uranium and radium on stream sediments near the mine dewatering outfall. Analysis of fish skeletons indicated radium uptake corresponding to dissolved radium-226 concentration exceeding 1 pCi/ℓ. Resampling in 1970 showed a decrease in radioactivity values in sediment but a tenfold increase in fish uptake of uranium relative to other fish populations in the basin. Radioactivity concentrations in fish tissue were highest near the mine effluent outfalls but did not constitute a major source of radioactive intake by consumers.

In June 1971, the EPA Radiological Activities Section of Region VIII (Denver) made a field reconnaissance of uranium mining and milling activities in the Shirley Basin area. Radiological analyses of water and sediment samples in the Shirley Basin and in the Bates Hole drainage basin to the north unquestionably indicated significant increases in radioactivity levels in water, sediment, and fish because of effluent discharge from mines and mill tailings. Concentrations of dissolved radium-226 and uranium in mine effluent were well above background. The discharges were not considered a source of radiation dose to the populace (residents and transients) because of remoteness and lack of water use, but toxic effects on fish were of concern (M. Lammering, written communications, 1979). Monitoring in 1972 by the Wyoming Game and Fish Department showed water quality effects as far as seven miles downstream. From 1970 to 1972, radium-226 concentrations remained stable, but uranium increased. Fish samples collected in 1972 showed increased amounts of radium in the flesh compared to the 1970 results. Soil samples from a creek that received mine effluent indicated relatively large transport and enrichment of uranium and radium. Radium in particular was enriched in the sediments and showed temporal variations indicative of successive scouring and removal, presumably in flood flows. Precipitation of uranium compounds was not apparent, probably because oxidized uranyl species are quite soluble in natural water.

To further our understanding of the role aqueous pathways play in contaminant dispersal, we monitored stable and radioactive trace elements in the Spring of 1979 in surface runoff from ore, sub-ore, and overburden piles from the Morton Ranch area of active mining in Wyoming. EPA personnel selected
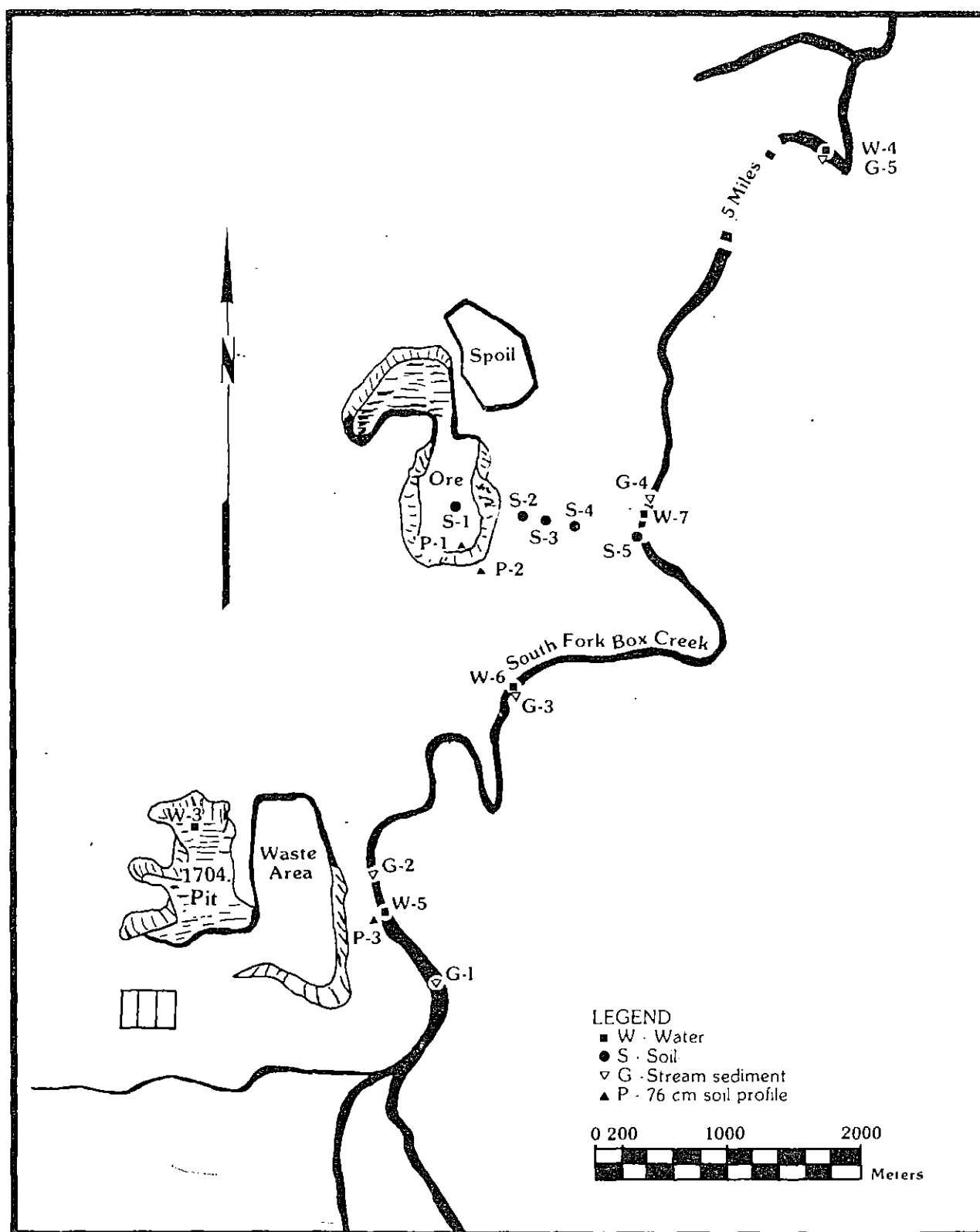
the basic study areas and assisted in sample collecting. Most of the sampling, analysis, and interpretation was done under contract with Battelle, Pacific Northwest Laboratories and is the subject of a draft report (Wo79). Appendix G contains a more complete discussion and listing of the data. Since runoff samples were unavailable, the sampling program emphasized the collection of soil samples in well-defined runoff gullies originating at surface stockpiles of minerals. Where the drainage systems intersected flowing water, upstream and downstream samples were collected. Most of the samples consisted of the top five centimeters of soil in the bottom of the drainage channels and three core profiles. Samples of the source material were also collected.

Trace element and radionuclide analyses of runoff from the Morton Ranch area are primarly based on surface sediments and vertical sediment profiles from the dry stream beds, since few streams or other forms of runoff were encountered. Figure 3.4 shows the waterways surrounding the inactive 1601 pit area and the semi-active 1704 pit area.

Three vertical soil profile samples were collected, two in an erosional drainage area from the ore and waste pits south of the 1601 pit and one in an erosional drainage bed on the east side of the waste pile of the 1704 pit. The radionuclide and chemical constituents of these samples along with analyses of other soil samples are reported in Appendix G. The results indicate aqueous leaching based on the radium/uranium ratios of about 12 in redeposited material in the alluvial fan area of the drainage, compared to a corresponding ratio of 0.9 in the undisturbed (sub-surface) material in the top alluvium profile.

Trace element data also indicate limited transport of mine contamination with respect to uranium, and to a lesser extent, Se and V. The profile samples containing 140 and 21 ppm U resulted from material transported from the adjacent ore piles. The aqueous samples similarly showed no unusual characteristics indicative of mine wastes.

An aqueous sample and the soil profiles collected near the 1704 pit similarly showed no evidence of mine-related pollution or leaching of uranium. This observation is based on the water and particulate analyses and the radionuclide analyses. These samples constitute a worst case, since the sediment samples were collected in the redeposited material of the waste pile drainage and should show greater levels of pollutants there than would fin-

Figure 3.4 Location of mines, ore and waste storage areas and monitoring stations at the Morton Ranch mine, South Powder River Basin, Wyoming.

ally reach the South Fork Creek bed.

Pollutant releases from the Morton Ranch, Wyoming uranium mining operations were not observable in water drainages of the surrounding area. The only significant movement of mine-related wastes was the transport of the stockpiled ore in erosional drainage areas on and immediately adjacent to the waste pile of the 1601 pit. Long-distance transport of these pollutants (primarily uranium) into the South Fork of Box Creek was not observable. The strongest evidence that mine wastes are a source of local soil and water contamination is the radiochemical data, and uranium in particular. Pollutant transport is almost entirely confined to the immediate area of the mines, although there has been some dispersal via water in the ephemeral streams. There is considerable disequilibrium between radium and uranium which may indicate leaching and remobilization of uranium. The possibility of natural disequilibrium in the ore body should not be overlooked.

3.2.3.2.3 Texas

A very comprehensive field and literature survey of elements associated with uranium deposits in south Texas (He79) revealed high to very high concentrations of molybdenum, arsenic, and selenium in areas of shallow mineralization; drainages adjacent to older, abandoned mines; and in some reclaimed areas. Areas of shallow mineralization have concentrations of several tens of ppm molybdenum and arsenic and up to 14 ppm selenium. Near surface material exposed by mining may have several hundred ppm molybdenum and arsenic. Waterborne transport of suspended or dissolved solids away from open pit mines resulted from mine water discharge and (or) surface runoff and erosion of abandoned spoil piles. Molybdenum from the mining areas could potentially aggravate natural soil problems leading to molybdenosis (Kab79). Additional careful study is suggested, particularly of areas receiving mine drainage as pumped water or overland flow.

Lakes or ponds associated with 10 mine locations in Karnes and Live Oak counties contained water unsuitable for drinking without prior treatment (It75). Generally, mineralization was also excessive and rendered the water unfit for irrigation. Air and terrestrial sampling revealed no health hazards from mining wastes and mined lands, but insects and other bottom fauna in the lakes concentrated radium-226 400 to 800 times the water concentration based on dry weight of the organisms.

3.2.3.2.4 New Mexico

The principal investigations of the influence of uranium mining on water quality includes EPA and contracted (Wo79) work by the U.S. Environmental Protection Agency, the Department of Interior (Ku79), and ongoing studies by the State of New Mexico (J. Dudley, oral communication, 1979). Because of the co-location of mining and milling facilities, it is difficult to identify impacts from one versus the other.

Survey of groundwater and surface water quality in close proximity to the Jackpile-Paguate, Ambrosia Lake, and Churchrock mining areas (EPA75) revealed extensive discharges of mine water to the ambient environment, use of unlined ponds for settling suspended solids from mine dewatering, use of contaminated mine water as a potable supply (one facility), and failure of all facilities discharging to streams to have a valid NPDES permit. The volume of mine discharge, particularly in the Churchrock area and from a mine near Mount Taylor, led to use of the water for irrigation and stock. In other areas of Ambrosia Lake and near Churchrock, infiltration of mine water mixed with seepage from mill tailings ponds is causing local contamination of shallow, potable aquifers, but the problem is not considered serious and ongoing studies are underway. The State of New Mexico has installed a monitoring well network to determine temporal and spatial trends in groundwater quality. The U.S. Geological Survey, in particular, is monitoring surface flows and water quality in the Ambrosia Lake and Churchrock areas.

As part of the San Juan Basin Regional Uranium Study, the Department of Interior (DOI79) assisted by the U.S. Geological Survey (Ku79) examined selected water quality impacts from mining and milling and concluded that much of the mine effluent is suitable for irrigation, stock, and industrial use. Locally, it supports aquatic life and wildlife. Additional data on stream sediments are needed to evaluate the impact on water resources of erosion of waste rock from mines and mill tailings. It is preliminarily suggested (Ku79) that such erosion may be difficult to detect at distances of more than a few miles from the source because of the large amount of (natural) regional soil erosion. The results of the study are presently in draft form and may therefore be revised.

As part of the present study on uranium mining wastes, two New Mexico areas containing inactive mines were surveyed in the Spring of 1979. Stable and radioactive trace elements were monitored in surface runoff from sub-ore

and overburden piles in Ambrosia Lake and in the nearby Poison Canyon area. EPA staff selected the basic study areas and assisted in sample collection. The bulk of the sampling, analytical, and interpretation phases was done by Battelle, Pacific Northwest Laboratories (Wo79). Appendix G contains the data and discussion. As in the case of the Wyoming study area, the sampling program emphasized stream sediment sampling, cores, and shallow (5 cm thick) grab samples at the land surface. Samples of the source material were also collected. Figures 3.5 and 3.6 show the location of the study areas and sampling stations in New Mexico. Samples of the source material were col-lected at one of the two New Mexico drainage systems investigated. One system in Poison Canyon, New Mexico, was adjacent to several small surface operations as well as an underground mine site. For this system, no single source could be defined for the runoff constituents.

The Poison Canyon mine drainage system is a dry creek bed. The course of this creek passes an abandoned underground mine site from which it can receive runoff water. It then passes through a dirt roadway and follows a course adjacent to some small open pit mines. After a distance of several hundred kilometers, it joins a second branch drainage that originates next to a waste pile from one of the open pit mines. Samples were collected along this waterway starting with a background sample upstream of the underground mine about 200 m from the road. The first downstream sample was collected about 130 m downstream from the roadway. This was upstream of the runoff source originating in the open pit mine. The remaining samples were col-lected along the drainage way (Fig. 3.5), below contamination sources from the open pit operations.

A second site, the San Mateo Mine and environs, is located in the south-east portion of the Ambrosia Lake mining district. Large mine waste piles, a heap leaching operation, and a mine drainage pond are prominent at the site, which drains northward to San Mateo Creek.

Soil composites were collected at the waste pile and heap leach pile. These represent the source term for possible contamination of the watershed. The drainage samples were collected following one channel down the waste pile face to the intersection with San Mateo Creek, which was followed for a distance of 500 to 600 m from the site. Additional samples were collected in the gullies leading from the heap leach area and one of the off-site gullies. The latter represents blank soil upstream of the drainage water. Sampling sites are noted in Fig. 3.6. No significant contamination from the under-
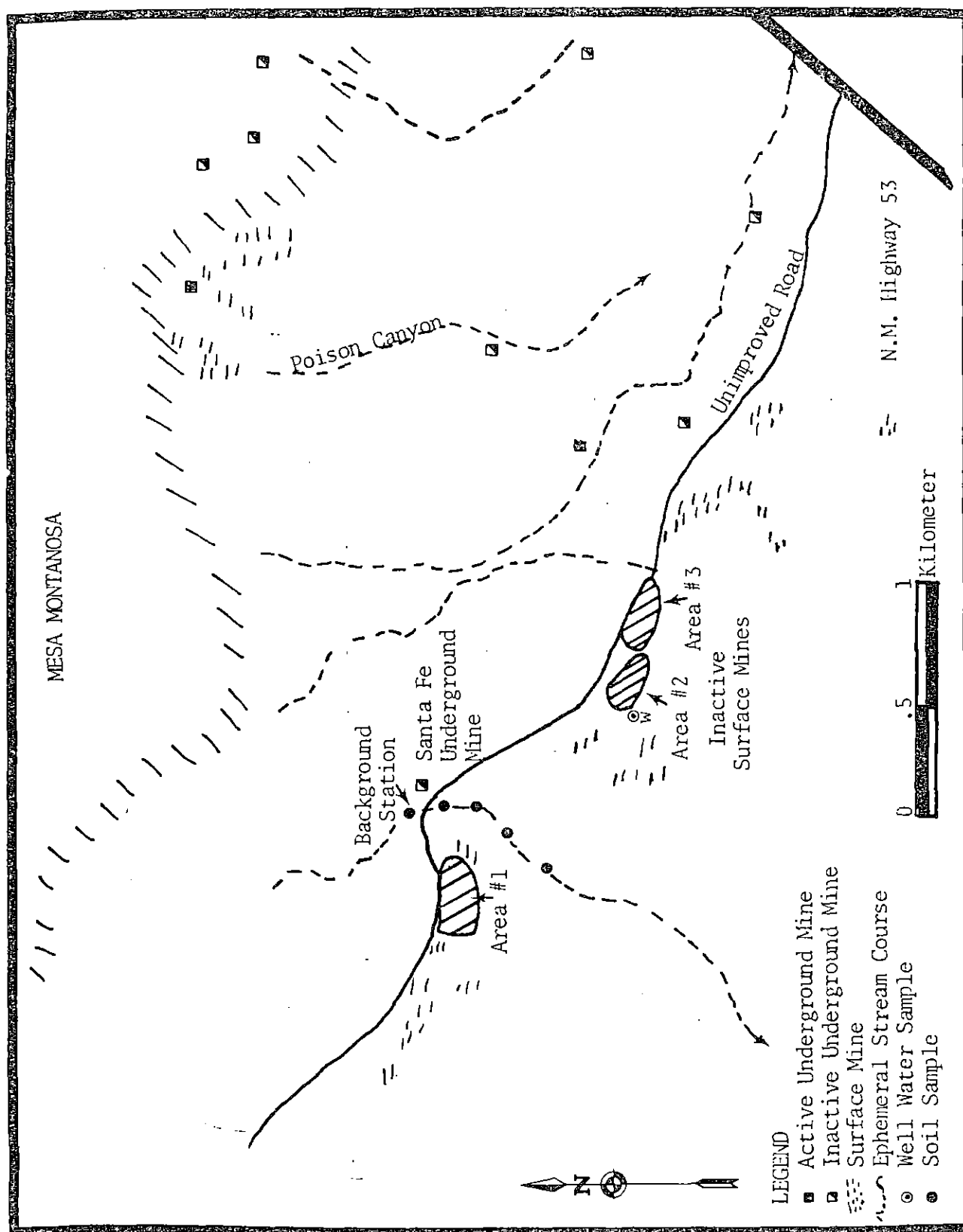
Figure 3.5 Location of study areas, sampling stations and uranium mines, Poison Canyon area, McKinley County, New Mexico.
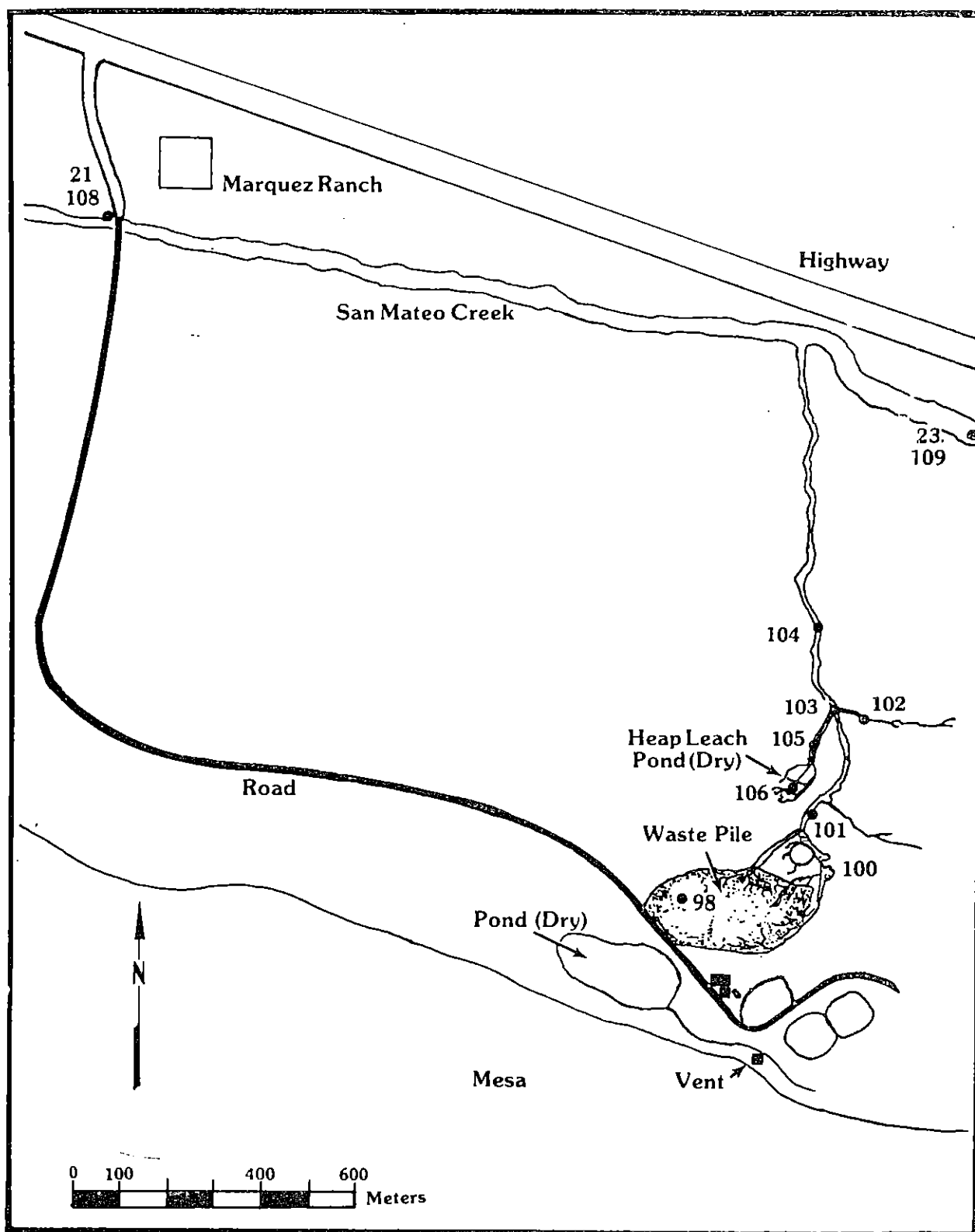
Figure 3.6 Sample locations for radionuclides and select trace metals in sediments, San Mateo mine, New Mexico.

ground site was detected. The Ra-226 content of the soil was about two times the background level at the furthest downstream site. This sample was collected about 130 m from the apparent source.

In summary, at the New Mexico inactive mine drainages the most prominent indicator of runoff from above-ground mineral storage is radium-226 in stream bed sediments. Concentrations in the source material are almost two orders of magnitude higher than those measured in the background soils. Elements such as uranium and selenium also have as large a concentration gradient, with concentrations decreasing downstream. At Poison Canyon, the radium-226 concentration diminished to two times background in a distance of approximately 100 m, while at the San Mateo site the distance was about 400 to 500 m. This may reflect either a more rapid transport by faster flowing water at the San Mateo site or, more likely, the larger source term there relative to background. At the San Mateo mine, radium-226 concentrations in water and sediments are significantly elevated downstream relative to upstream of the mine drainage.

3.2.3.3 Summary

The field studies conducted to date on the impacts of uranium mining on water quality are somewhat contradictory. Although no cases of gross, widespread contamination of groundwater or streams can be documented for uranium mining, there are cases of local contamination of water and sediments. From standpoints of theory and field data, there is need for cautious optimism in the use of local soil and water resources as sinks for waste discharge. Although numerous studies indicate that considerable reliance can be placed on the various physical and chemical processes to protect natural waters from contamination, investigations generally warn against using such studies to predict what may happen in other situations (Ru76; NRC79b; Ku79; Fu77; Am78). Laboratory results are highly dependent on the chemical properties of the fluid matrix and the physical and chemical properties of the particular soil studied. Results of field studies are site and time specific and have often suffered from inconsistent and undefined sampling and sample preservation techniques and questionable analytical measurements (Ku79; Ha78; Si77).

Our analyses reveal that there have been local water quality problems from mine water and wastes. Although widespread hazards have not been identified, this may be false security insofar as the present status of knowledge concerning trace element mobility in aqueous settings representative of

uranium mining areas is rather unclear from both theoretical and real-data standpoints. Most often, effects of mining are interspersed with and masked by impacts from uranium milling. This complicates or renders impossible any meaningful interpretation of the mining-related data. Despite the attempt to sort out some of the information on trace element mobility, there is insufficient understanding at this time to dismiss or otherwise reduce the significance of trace element contributions (from mining activities) to surface streams and, to a lesser extent, to groundwater.

We conclude that there is considerable information on the topics of trace element chemistry. It is also clear that trace element concentrations in natural fresh water are highly variable on both macro and micro geographic scales. There is great difficulty in correlating concentrations with such characteristics as streamflow or lithologic environment. Accurate prediction of the behavior and cycling of trace elements through water and sediments first involves characterization of physical states such as particle size and form (chelate, colloid, complex ion, precipitate, etc.), speciation, and availability to plants and animals (An73). Andelman concludes "...that there can be large differences in trace element concentrations [in water], on both a macro and micro geographic scale, and that such variations often occur in an unsystematic and nonpredictable fashion."

We recommend additional studies of spatial and temporal variations, sources and sinks of trace elements, chemical interactions within the hydrogeologic system, interactions between surface and groundwater systems, effects on aquatic biota, and effects on water use (human consumption, stock watering, irrigation). Periodic monitoring in certain areas would allow for the detection of the long-term trends of potential changes that would accompany anticipated increases in future mining activity -- during a period of increased competition for scarce water resources.

3.3 Surface Mining

3.3.1 Solid Wastes

Surface mining consists of removing materials, separating them into ore, sub-ore, and overburden, and storing them in separate piles on the surface near the mine for various periods of time (Section 1.3.2). The various storage piles are managed differently, vary in size and level of contaminants, and exist for varying periods of time. All are potential sources of

contamination to the environment via dusts suspended and transported by the wind, precipitation runoff, and Rn-222 emanation (Fig. 3.7).

3.3.1.1 Overburden Piles

Surface mining produces spoils at a rate of millions to tens-of-millions of tons per year. Unless this material is used to backfill the pit, large surface areas -- 40 hectares to over 400 hectares -- are covered to depths varying from a few meters to over 100 meters (Ka75, NRC77a, NRC77b, DOA78, Pe79).

Most of the mines begun since the early to mid 1970's use overburden to backfill mined-out areas of the pit (Ka75). Since older mines usually did not, erosion of their storage piles by water and wind may present an environmental problem (Ka75). In addition, the large amounts of overburden that past and present mines have used for road and dike construction and backfill also may present an environmental problem.

The annual average ore production of the 63 surface mines operating in the United States in 1978 was 1.2×10^5 MT (Section 1.3.1). Assuming an overburden to ore ratio of 50:1 (Section 1.3.2), the average annual production of overburden was about 6.0×10^6 MT per mine. A recent study of the eight large mines that accounted for 68 percent of the total 1977 United States U_3O_8 production from surface mines recommends the following average production parameters (Ni79):

1. ore production = 5.1×10^5 MT/yr
2. average ore grade = 0.11 percent U_3O_8
3. overburden:ore ratio = 77
4. overburden production = 4.0×10^7 MT/yr
5. mining days/yr = 330 d/yr

Surface areas of hypothetical overburden piles were computed using the above 63-mine and 8-mine overburden production rates and the following assumptions:

1. an average density of 2.0 MT/m^3 - reported values vary between 1.6 and 2.7 (Ro78, DOA78, NRC78a, Ni79)
2. the dumps are on level terrain
3. a rectangular waste dump with the length twice the width, and sides that slope at 45° angles (Fig. 3.8a)
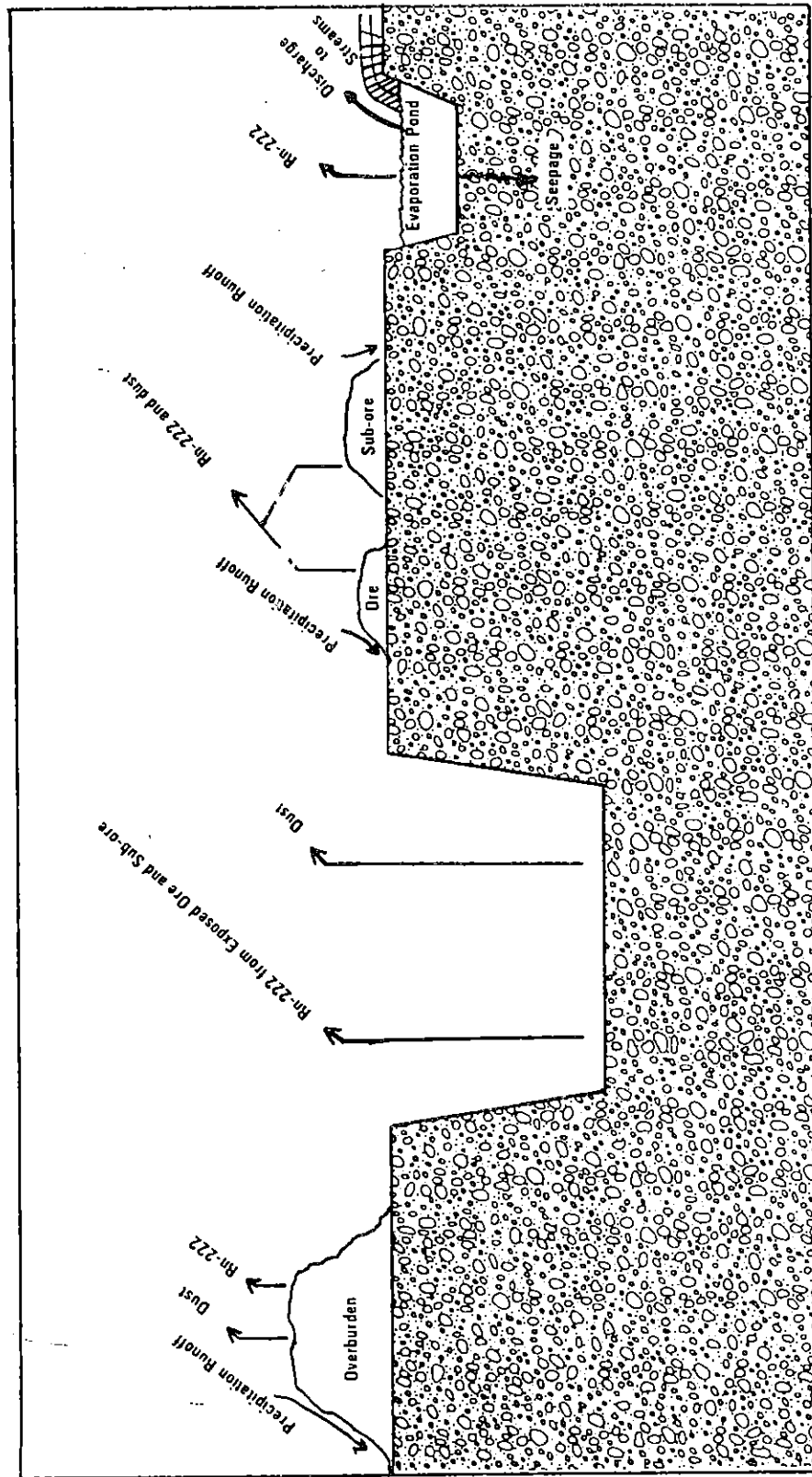
Figure 3.7 Potential sources of environmental contamination from active open pit uranium mines.

4. a waste dump in the shape of a truncated right-circular cone with 45° angled sides (Fig. 3.8b)

5. a bulking factor of 25% or 1.25 (Burris, E., Navajo Engineering and Construction Authority, Shiprock, NM, 2/80 personal communication)

Table 3.11 lists the surface areas of the hypothetical overburden piles in the following three cases:

Case 1 - one year production with no backfilling

Case 2 - backfilling concurrent with mining - assumes 7 pits opened in a 17-year mine life with overburden from each successively mined pit used to backfill a previously completed pit, resulting in an equivalent of one pit of overburden (2.4-yr production) stored on the surface (Ni79)

Case 3 - no backfilling during the 17-year mine life

The quantities of dust and Rn-222 that become airborne are directly proportional to the surface areas of waste piles. Table 3.11 shows the large variations possible between surface areas of waste piles at some active mines. Waste piles also cover various areas of terrain. However, for the same volumes, there are no significant differences in surface area or area of terrain covered for the two configurations of waste piles used in this study. Case 2 approximates recently activated mines, and Case 3 approximates older mines.

The type of rock in overburden spoil piles depends on the locations of the ore zones. Common rock types of New Mexico, Wyoming, and Texas mines include sandstone, claystone, siltstone, shale, and limestone, and unconsolidated silt, gravel, and sand (Co78, Pe79, Wy77, Ri78). In Texas, there are also lignite beds, tuffaceous silts, and some nearly pure volcanic ash
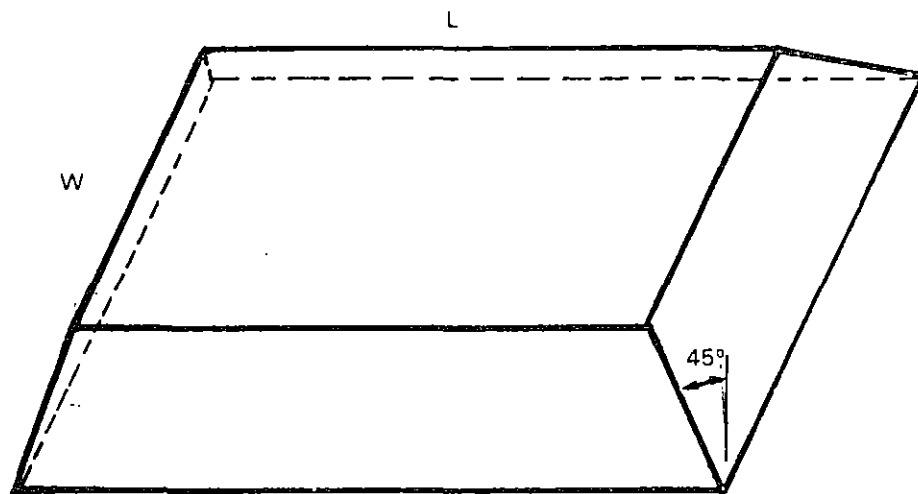
a) A rectangular pile with length twice the width and 45 degree sloping sides
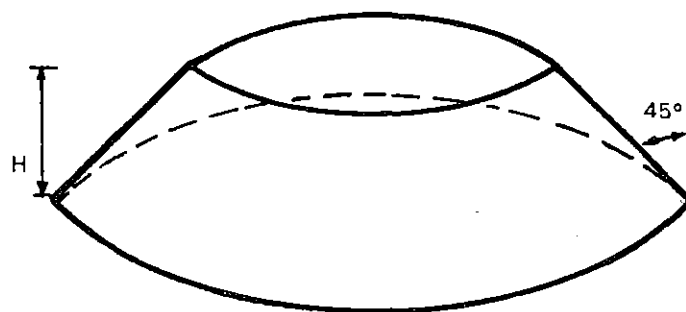
b) A frustum of a regular cone with 45 degree sloping sides

Figure 3.8 Storage pile configurations assumed at surface and underground mines.

(Ka75). Coal veins are often present in Wyoming and New Mexico (Wy77, Ri78). However, the most abundant material in waste rock dumps will probably be clastic sedimentary rocks: sandstone, siltstone, and shale.

There is great variation in the particle size of material in waste piles, and this variation is important. Large particles (>30μm)*, because they usually settle within a few hundred feet of their origin, do not con-. tribute to the airborne dust concentration (EPA77b). The potential for human respiration of the wind suspended dusts is also strongly influenced by the mean particle diameter (ICRP66).

Overburden rock is as large as available equipment can load and haul to the storage area. Rocks too large to handle with available equipment are broken into manageable sizes by small, explosive charges. Hence, rock particles will vary from less than a μm to a meter or more in diameter. Since weathering eventually breaks down the larger stones, the fraction of smaller particles increases over time.

Particle size distributions of material in waste rock piles at uranium mines have not been determined. It is likely that this material has a greater fraction of larger particles than that associated with crushed uranium mill tailings. Table 3.12 shows an example of the particle size distribution in the latter and the mean particle size distribution from a study of shale overburden removed from a surface mine in Pennsylvania (Ro78). Although the distribution fractions differ, a gross comparison can be made between the particle size of mill tailings and overburden waste. About 28 percent of the tailings were less than 50 μm in diameter, and only about 12 percent of the particles in the overburden pile had similarly small diameters. Because only particles smaller than 30 μm are likely to remain suspended by the wind for any significant distance (EPA77b), probably less than 10 percent of the overburden is a potential source of environmental contamination via wind erosion.

Table 3.13 shows the natural radionuclide concentrations in common rock types in the United States. In sedimentary rocks, which are common in the major uranium mining regions, the U-238 concentrations vary from less than 1 ppm** to about 4 ppm. Natural radioactivity usually is somewhat higher in the western states, and the uranium content in overburden prior to mining

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*μm = micrometer = 10^{-6} meters.

**ppm = parts-per-million = 10^{-6} grams per gram of rock.

Table 3.11   Estimated surface areas associated with overburden piles

Management[a]	Pile Height, m	Overburden Volume[b], m^3	Surface Area of Pile, m^2	Terrain Covered, Hectares
Rectangular Pile[c]				
		Average Large Mine[d]		
Case 1	65	2.5×10^7	5.2×10^5	48
Case 2	65	6.0×10^7	1.1×10^6	106
Case 2	30	6.0×10^7	2.2×10^6	209
Case 3	65	4.2×10^8	7.1×10^6	682
		Average Mine[e]		
Case 1	65	3.8×10^6	1.0×10^5	10
Case 2	65	9.0×10^6	2.2×10^5	20
Case 2	30	9.0×10^6	3.6×10^5	34
Case 3	65	6.4×10^7	1.2×10^6	113
Truncated Cone[f]				
		Average Large Mine[d]		
Case 1	65	2.5×10^7	5.2×10^5	46
Case 2	65	6.0×10^7	1.1×10^6	104
Case 2	30	6.0×10^7	2.1×10^6	208
Case 3	65	4.2×10^8	7.1×10^6	683
		Average Mine[e]		
Case 1	65	3.8×10^6	1.1×10^5	9
Case 2	65	9.0×10^6	2.2×10^5	18
Case 2	30	9.0×10^6	3.5×10^5	33
Case 3	65	6.4×10^7	1.2×10^6	110

Table 3.11 (continued)

(a)Management:

    Case 1 - one year production with no backfilling

    Case 2 - backfilling concurrent with mining - assumes 7 pits
              opened in a 17-year mine life and equivalent of one-
              pit overburden (2.4 year production) remains on surface

    Case 3 - no backfilling during 17-year mine life

(b)Volume = production (MT/yr) x production years x bulking factor (1.25)
    ÷ by density (2.0 MT/m^3).

(c)Length of pile is twice the width and the sides slope at a 45° angle
    (Fig. 3.8a).

(d)Overburden production = 4.0 x 10^7 MT/yr.

(e)Average 1978 overburden production of all 63 surface mines, assuming an
    overburden:ore ratio of 50/1, 6.0 x 10^6 MT/yr per mine.

(f)A frustum of a regular cone with 45° sloping sides (Fig. 3.8b).

is about 4 ppm (Ni79). However, during mining, some low-grade ore mixes with
the overburden and may increase the concentration of the pile to as high as
20 ppm U_3O_8 (Ni79). This is equivalent to 12.6 disintegrations per minute
(dpm) per gram of overburden. The progeny of the uranium will contribute
additional radioactivity. Although there are local disequilibria between
U-238 and its principal daughters, Th-230 and Ra-226, in ore-bearing rock,
secular equilibrium will be assumed (Wo79). Small quantities of Th-232 and
progeny will provide additional radioactivity. There is no apparent rela-
tionship between the Th-232 and U-238 decay chains. Th-232 concentrations in
ores and host rock range from less than a pCi/g to a few pCi/g regardless of
the U-238 concentration (Wo79).

Table 3.12   Particle size distributions of mill tailings
and mine overburden

Mill Tailings[a]			Overburden[b]	
Particle Size, μm	Weight Percent	Conc.[c] Avg. Conc.	Particle Size, μm	Weight Percent
250	60.3	0.15	>2000	75
125-250	7.5	0.03		
53-125	4.2	0.03	50-2000	13
44-53	3.8	0.03		
20-44	7.8	0.75	2-50	8
7-20	7.2	1.5		
1.4-7	9.1	4.6	<2	4
<1.4	0.0			

[a] Source: Sc79.

[b] Source: Ro78.

[c] The concentration of radionuclides in that fraction divided by the average concentration.

Table 3.13   Natural radionuclide concentrations in various
common rock types

Rock Type	U-238		Th-232		K-40	
	ppm	pCi/g	ppm	pCi/g	ppm	pCi/g
Igneous						
Basic	0.9	0.3	2.7	0.3	1.2	8.4
Granite	4.7	1.6	20	2.2	5.0	35
Sedimentary						
Shale	3.7	1.2	12	1.3	3.2	22
Sandstone	0.45	0.15	1.7	0.2	1.1	7.7
Limestone	2.2	0.7	1.7	0.2	0.32	2.2

Source: Oa72.

Table 3.14 shows the results of an extended airborne particle sampling program near a surface mine in New Mexico (Ea79). Although the on-site source of the radioactivity measured on these filters is undetermined, ore and sub-ore piles, waste rock piles, and mining activity all probably contribute. The higher activities reflect a greater contribution from ore dusts. From these air measurements, the above assumed average uranium concentration in overburden, 12.6 dpm/g ($\cong 6$ pCi/g), appears reasonable. These data also indicate that the progeny of U-238 through Ra-226 are in near secular equilibrium. The Th-232 concentration is about 1 pCi/g and, as indicated above, independent of the uranium concentration. Considering all available data, the radioactive source terms for overburden piles will be as follows: (1) U-238 and progeny = 6 pCi/g (0.0020 percent U_3O_8; (2) activity ratio (dust:overburden) = 2.5 (Section 3.3.1.2); and (3) Th-232 and progeny = 1 pCi/g. Figures 3.1 and 3.2 show the uranium and thorium decay series.

Table 3.14 Annual average airborne radionuclide concentrations in the vicinity of an open pit uranium mine, pCi/g

Location	U-238	Th-230	Ra-226	Th-232	U-238/Th-232
Jackpile Housing	76	80	70	1.2	63
Paguate	13	12	13	1.3	10
Bibo	9	7	5	1.3	7
Mesita	3	2	3	0.7	4
Old Laguna[a]	5	2	3	0.4	13

[a]Background location.

Source: Ea79.

Little information is available on stable element concentrations in overburden rock. Table 3.15 summarizes the analyses of a few grab samples of soil and rock from a uranium mine in New Mexico and one in Wyoming (Wo79). Except for possibly Se, V, and As, there are no significant concentrations of stable elements attributable to uranium mining. Considering the typically high natural Se and V contents of many minerals common to these areas and the limited number of analyses, the inference of pollution is indefinite. A relationship between uranium and the stable element concentrations does not appear to exist. Thus, the stable element concentrations in overburden from the model surface mine will be the average concentrations of samples 6, 7, and 8 in Table 3.15. Table 3.16 lists the average concentrations.

3.3.1.2 Ore Stockpiles

Ore is often stockpiled at the mine as well as the mill. Although ore stockpiles are much smaller than the overburden waste piles, the concentrations of most radioactive contaminants are much greater in ore-bearing rock than in overburden. In addition, ore is stockpiled at the mine for shorter periods of time than waste rock. Ore stockpile residence times vary from mine to mine and range from a few days to a few months. The recent study of 8 large surface mines cited 41 days as an average ore stockpile residence time (Ni79). We will use this value to estimate the average area of ore stockpiles.

The average of the 63 operating surface mines produced 1.2×10^5 MT of ore during 1978. Assuming 330 working days per year and a 41-day ore stockpile residence time, a 1.5×10^4 MT ore stockpile would exist at the average mine. In comparison, the recent Battelle study reported that the average of eight large surface mines produced 1550 MT of ore per day, which would yield a 6.3×10^4 MT ore storage pile, assuming the same residence time (Ni79). The ore piles vary in height at different mines and different times. One study reports a maximum pile height of 9.2 m (30 ft) (Ni79), and at another site the maximum and equilibrium ore pile heights are estimated to be 6.7 m and 3.1 m, respectively (NRC78a). Using these parameters and a bulking factor of 1.25 (Burris, E., Navajo Engineering and Construction Authority, Shiprock, NM, 2/80, personal communciation), the pile surface and pad areas were computed for the two production rates and two pile heights, 9.2 m and

Table 3.15 Uranium and stable element concentrations measured in rock and soil samples from two uranium mines

Sample	As	Ba	Cu	Cr	Fe[a]	Hg	K[a]	Mn	Mo	Pb	Se	Sr	V	Zn	U
Wyoming															
1. Top Soil Piles	3.2	700	13	46	1.3	<4	2.2	190	2.9	23	<1	89	60	37	6
2. Sub-ore	<1.8	6800	9	<36	1.2	10	2.3	140	<2.2	22	2.1	128	<100	25	61
3. Ore	5.4	800	9	<27	1.1	<7	2.3	180	<2.9	16	28	94	200	25	370
New Mexico															
4. Background Soil	4.1	450	12	<23	0.9	<4	1.8	200	5.5	12	<1	72	<50	22	<5
5. Background Soil	2.3	440	9	<20	0.8	<4	1.6	190	4.9	13	<1	50	<50	19	<5
6. Waste Pile	7.8	540	11	<28	0.8	<5	1.4	260	2.5	10	<1	99	<70	23	8
7. Waste Pile	14	280	21	<43	0.7	<8	0.5	750	<2.8	31	3.1	178	180	23	189
8. Sub-ore + waste	4.1	45	22	<51	0.3	<6	0.1	446	<1.8	25	<1.4	179	<55	13	57
9. Ore	6.0	64	27	<48	0.4	<6	0.2	673	<1.8	31	1.5	323	<55	14	---

Concentration, µg/g

Source: Wo79.
[a] Units are percent.

Table 3.16   Concentration of radionuclides (pCi/g) and stable elements
(μg/g) in overburden rock from the model surface mines

Element	Concentration	Element	Concentration
Arsenic	9	Selenium	2
Barium	290	Strontium	150
Copper	18	Vanadium	100
Chromium	<51	Zinc	20
Iron[a]	0.6	U-238	6
Mercury	<8	Th-230	6
Potassium[a]	0.7	Ra-226	6
Manganese	485	Pb-210	6
Molybdenum	2.5	Po-210	6
Lead	22	Th-232	1

[a] Units are percent.

3.1 m, assuming the same geometric configurations as for the overburden piles
(Fig. 3.8).   Table 3.17 gives the results. The computed surface areas of an
average ore stockpile vary with volume of ore stored and pile height, but
they are relatively independent of the pile shape.

Uranium deposits exist in sedimentary, metamorphic, and igneous for-
mations. Sedimentary formations, primarily sandstone, siltstone, mudstone,
and limestone generally host stratiform ore deposits often accompanied by
carbonaceous material.   Vein-type deposits usually occur in fractures of
igneous and metamorphic formations.   In the Rocky Mountain mining regions,
about 98 percent of the recovered U_3O_8 comes from sandstone and related-type
rock (St78).   Sedimentary formations, principally sandstone, have been the
predominant host for uranium in South Texas (Ka75).

Table 3.17 Estimated average areas of ore pile surface and pad

Pile Configuration[a]	Pile Height, m	Surface Area of Pile, m^2	Ore Pad Area, m^2
Average Large Mine[b]			
Rectangular	9.2	6,300	5,700
Truncated Cone	9.2	6,200	5,300
Rectangular	3.1	14,000	13,500
Truncated Cone	3.1	13,700	13,200
Average Mine[c]			
Rectangular	9.2	1,860	1,820
Truncated Cone	9.2	2,000	1,580
Rectangular	3.1	3,660	3,420
Truncated Cone	3.1	3,590	3,340

[a] See Figure 3.8.

[b] Volume of ore = 6.3×10^4 MT (41 day production) x 1.25 (bulking factor) ÷ 2.0 MT/m^3 = 3.9×10^4 m^3.

[c] Volume of ore = 1.5×10^4 MT (41 day production) x 1.25 (bulking factor) ÷ 2.0 MT/m^3 = $9.4 \times 10^3 m^3$.

The DOE does not expect the mineralogical characteristics of uranium ore to change appreciably in the future, since the known reserves are mainly in sandstone or a related host (DOE79). This fact is apparent from the data in Table 3.18, which gives the distribution of ore reserves in the United States by type of host rock. More than 97 percent of the uranium reserves are in sedimentary formations, primarily sandstone. Hence, it is reasonable to assume that ore stockpiles in the future will continue to consist mainly of a friable (easily crumbled) sandstone rock.

No data are presently available on the particle size distribution of material in ore stockpiles. Thus, the particle size distribution of ore will be assumed to be similar to that of overburden rock.

The average grade of ore mined in 1978 was about 0.14 percent U_3O_8, but this will decline in future years (DOE79). The average grades of ore associated with tne $30 and $50 reserves are 0.10 percent and 0.07 percent U_3O_8, respectively (DOE79). Assuming the average grade of ore mined in the next decade to be about 0.10 percent U_3O_8, the average uranium concentration in ore stockpiles will be 285 pCi/g (632 dpm/g). Although secular equilibrium in the uranium decay chain may not totally exist in some cases due to leaching by groundwater with subsequent redeposition, it appears reasonable to assume that radioactive equilibrium exists in a general assessment.

As discussed earlier, ambient Th-232 concentrations in the vicinity of a uranium mine range between 1 to 2 pCi/g. However, a concentration of 0.01 percent thorium is typical for ore from some surface mines (Mi76). This concentration is equivalent to 11 pCi Th-232/g of ore.

Uranium occurs in many ores as a secondary deposition. In a reducing environment, the soluble uranyl ion converts to insoluble uranium oxide and deposits preferentially on the smaller particles. (The total surface area of a given mass of smaller particles is greater than for larger particles.) Therefore, dusts that consist primarily of small particles have a greater specific concentration than ore as a whole (Table 3.12). The common procedure for computing uranium concentration in dust is to multiply the average concentration in the ore by 2.5 (NRC77a, NRC78a).

Table 3.18 Distribution of ore reserves by the type of host

Host Type	MT of Ore (10^6)	MT of U_3O_8	Percent Total Tons, U_3O_8
Sedimentary[a]	1,143.2	810,000	97.1
Lignite Materials	2.2	3,000	0.4
Limestone	1.3	1,200	0.1
Igneous and Metamorphic	32.7	20,400	2.4
Totals	1,179.4	834,600	100.0

[a] Principally sandstone, but includes conglomerates, shale, mudstone, etc.

Note.--The reserves are $50 or less per pound U_3O_8, effective January 1, 1979 (DOE79).

Table 3.19 Average stable element concentrations in sandstone ores of New Mexico

Metal	Concentration, μg/g[a]	Metal	Concentration, μg/g[a]
Arsenic	86 (10-890)	Manganese	960 (70-3,000)
Barium	920 (150-1500)	Molybdenum	115 (3-700)
Cadmium	ND[b]	Nickel	20 (7-70)
Cobalt	16 (3-150)	Lead	78 (3-300)
Copper	61 (15-300)	Ruthenium	ND
Chromium	20 (7-70)	Selenium	110 (1-625)
Iron	15,700 (3,000-70,000)	Strontium	130 (1.5-300)
Mercury	ND	Vanadium	1410 (70-7,000)
Potassium	25,000 (7,000-30,000)	Zinc	29 (10-70)
Magnesium	3,500 (700-15,000)		

[a] Range of concentrations given in parentheses.

[b] ND - not detected

Note.--Ore samples are Dakota and Morrison sandstone from 25 uranium mines (Hi69).

In accord with the above discussion, we assume the following estimated average radionuclide source terms for ore stockpiles: (1) U-238 and progeny = 285 pCi/g ore (0.10 percent U_3O_8); (2) Activity ratio (dust:ore) = 2.5; and (3) Th-232 and progeny = 10 pCi/g ore.

Stable elements -- molybdenum, selenium, arsenic, manganese, vanadium, copper, zinc, and lead -- often associated with uranium ore at elevated concentrations may cause deleterious environmental and health effects. Mercury and cadmium are present only on rare occasions (Th78). However, as discussed above, there is no apparent relationship between concentration of stable elements and ore grade (Wo79). Table 3.19 lists measured (Hi69) concentrations of stable elements in 25 sandstone ores from New Mexico and average concentrations computed from these data. We assume the average concentration for the ore from the model surface mine.

3.3.1.3 Sub-ore Piles

All mines recover some rock containing uranium ore that at the time of mining is uneconomic to mill. The grade of this "sub-ore" varies with the "cutoff" level assigned by the mill. Some mines process sub-ore by heap leaching, which changes the chemical properties and constituents of the pile (Section 1.3.5.1). However, most mines store the sub-ore in separate piles and recover it when it becomes economically feasible.

The sizes of sub-ore dump piles vary with the quantity of ore mined and its grade. One study suggests that the sub-ore accumulation rate equals the ore production rate (Ni79), a ratio similar to that reported for the Sweetwater uranium mining operation (NRC77a). Using this assumption with the ore production rates given above for the average large mine and average mine, 5.1 x 10^5 MT/yr and 1.2 x 10^5 MT/yr, respectively, the average sizes of sub-ore piles generated at a constant rate during the 17-year active life of a mine were based on an 8.5 year accumulation and a bulking factor of 1.25. Figure 3.8 shows the shapes of the piles assumed, and Table 3.20 gives the results for piles 30 m high. The surface areas of the two pile configurations differ very little.

The mineralogical characteristics of ore and sub-ore are very similar. Thus, the distribution in Table 3.18 will apply to sub-ore. This study considers the particle size distribution of sub-ore the same as for overburden and ore.

In the early mining years, the ore cutoff grade was usually about 0.15 percent U_3O_8. However, this has continually decreased until today the cutoff ore grade is about 0.03 percent U_3O_8 (Ni79, NRC77a). Hence the ore content of these piles will be less than 0.03 percent U_3O_8, and the average content has been estimated to be one-half the cutoff grade, or 0.015 percent U_3O_8 (Ni79), which is equivalent to 43 pCi U-238/g (95 dpm/g). Also, the uranium in the sub-ore, as in ore, is assumed to be in secular equilibrium with its progeny. Because the occurrence of uranium in sub-ore is the same as in ore and the mineralogies are similar, the uranium in sub-ore should be concentrated on small particles by the same factor as in ore, 2.5.

The Th-232 concentration in sub-ore is between the ambient level and that in the associated ore, 1 pCi/g to 11 pCi/g. For lack of measured Th-232 concentrations, we assume that less than 2 pCi/g of Th-232 will be present (Table 3.14). The radiological significance of an error in this assumption will be small.

From the above discussion, we assume the following estimated average radionuclide source terms for sub-ore piles: U-238 and progeny = 40 pCi/g (0.015 percent U_3O_8); activity ratio (dust:sub-ore) = 2.5; and Th-232 and progeny = 2 pCi/g. Figures 3.1 and 3.2 show the uranium and thorium progeny.

Table 3.20 Estimated average surface areas of sub-ore piles during the 17-year active mining period

Pile Configuration[a]	Surface Area of Pile, m^2	Terrain Covered, Hectares
Average Large Mine[b]		
Rectangular	1.2×10^5	11
Truncated Cone	1.2×10^5	11
Average Mine[c]		
Rectangular	3.5×10^4	3.2
Truncated Cone	3.6×10^4	3.0

[a]See Fig. 3.8.

[b]Volume of sub-ore = 8.5 yr x 5.1 x 10^5 MT/yr x 1.25 ÷ 2.0 MT/m^3 = 2.7 x 10^6 m^3.

[c]Volume of sub-ore = 8.5 yr x 1.2 x 10^5 MT/yr x 1.25 ÷ 2.0 MT/m^3 = 6.4 x $10^5 m^3$.

Stable elements observed in ore will also be present in sub-ore. Because stable element concentrations specific to sub-ore are unavailable and are unrelated to ore grade, concentrations in the sub-ore from the model surface mine will be assumed equal to those in the ore (Table 3.19).

3.3.1.4 Reclamation of Overburden Piles

Reclamation is usually done only for overburden piles. Ore stockpiles are continually being disturbed and their residence time is short. Also, sub-ore piles generally are not stabilized in anticipation of recovering the uranium at a later time. Hence, only overburden and waste rock piles are considered for stabilization and reclamation. Section 1.3.2 gives a brief description of these practices.

Backfilling mined out areas of the pit is necessary for an adequate reclamation program. Because of the swelling of earthern material once mined, sufficient material should be available to completely fill the pit when mining is completed. However, even though backfilling is generally being performed at most recently active mine sites, sufficient overburden is often not replaced to eliminate the pit.

Improperly stabilized spoil piles may become sources of contaminants to the environment. The wind can suspend and transport small-sized particles containing elevated levels of contaminants. Radon-222, produced by the radioactive decay of Ra-226 contained in the rocks, can emanate from the pile surfaces. Precipitation runoff from the piles can carry particulate matter and dissolved contaminants into the natural surface drainage system if rainfall exceeds the infiltration and holding capacity of the pile. The general procedure for reducing wind and water erosion is to grade the piles to conform to the natural terrain, cover the area with a layer of topsoil, and seed it with a native grass.

These spoils consist of unweathered and unconsolidated rock, coarse gravels, and sands and allied materials isolated from the natural processes that occur on surface soils. Consequently, spoils have poor textural properties and low water-holding capacities. Having no established flora to aerate the surface and make nutrients available, spoils are barren of nutrients required for plant growth. Hence, to sustain vegetation on these piles may be difficult because of poor soil quality and the arid conditions

in the principal mining regions. Therefore, all plant growth depends on the topsoil cover, which is generally less than 30 cm thick (Re76). This is often inadequate to store sufficient water and nutrients to sustain plant growth during extended dry periods. Soil irrigation and fertilization may be required for several years until plants can sustain themselves.

Proper grading of the spoil piles, with water management and conservation, can help reclamation. The piles should have less than a 3:1 slope to reduce surface water runoff and erosion (St78). Forming catchment basins and terraces to hold water on the spoils and reduce water erosion will also increase the amount of runoff available to the plants. It also has been determined that vegetation on north-facing slopes requires about half the applied water of that on south-facing slopes (Re76). Water requirements of vegetation on horizontal surfaces and east and west slopes are about intermediate between those of the north and south slopes. Hence, spoil piles with long, north slopes will conserve water and reduce the irrigation required. Locating piles on leeward slopes and away from natural drainage will also reduce wind and water erosion.

The reestablishment of native grasses and shrubs is essential for controlling wind and water erosion and providing wildlife habitat. Wyoming requires a pre-mining vegetation inventory for use in evaluating post-mining reclamation (Wy76). Similar statutes governing mine reclamation are in effect in other states (Section 1.4). The Soil Conservation Service has recommended seed mixtures that are best suited to climatic and soil conditions in different areas of the West (St78). Newly seeded areas are usually protected from grazing by fencing for at least two growing seasons to allow the plants to become established.

Abandoned pits fill with water and form small lakes that livestock and wildlife can use for drinking water, if the water is uncontaminated. But, unless properly managed, final pits may be hazards to people and wildlife. Therefore, steep walls should be graded to give safe access into the pit, and after grading, the pit banks should be seeded to minimize erosion and prevent the sides from sloughing off.

3.3.2   Mine Water Discharge

3.3.2.1   Data Sources
The principal sources of information used to model the mining region in

Wyoming are the site-specific EIS's and ER's for active and proposed mining/ milling operations and the NPDES permit data on discharge volume and quality. Several reports by state and federal agencies supplemented the foregoing, particularly with respect to estimating ambient water quality and flood volumes for various return periods and annual or monthly flows in principal streams of the region. Foremost among these is work by the State (Ha78), the U.S. Geological Survey (Cr78, Ho73), and the Soil Conservation Service (DOA75).

Self-monitoring data collected by industry and reported to EPA were also checked to ascertain compliance with NPDES permit conditions. Unfortunately, the permits do not specify limits on the volume of discharge; hence, the total mass or flux per unit of time may or may not agree with the values originally estimated by the discharges in the EIS, .ER, or license application.

3.3.2.2 Quantity and Quality of Discharge

The purpose of this section is to identify water quality associated with surface uranium mining in the Wyoming Basin. This area was selected for detailed source term characterization and pathways analysis because of past and ongoing uranium production, primarily by surface mining. A subsequent section (3.4.2) similarly addresses underground mining. The analysis to follow is incomplete and preliminary, owing to the limited existing data, the lack of opportunity for significant new investigations in the time of this study, and the decision to pursue the objectives on a "model area/model mine" approach. So many variables of ore occurrence, mining practices, climate, geology, and hydrology exist that a detailed investigation is unrealistic.

Table 3.21 summarizes water quality data for seven surface and three underground mines in Wyoming. Uranium averages 0.62 mg/ℓ and ranges from 0.02 to 1.3 mg/ℓ . Dissolved radium-226 is typically less than 4 pCi/ℓ , although one mine reportedly discharged 10.66 pCi/ℓ. Suspended solids average 24.9 mg/ℓ . There is considerable variation from one facility to another; the observed range is 2.7 to 87.2 mg/ℓ . Zinc is the only stable element consistently monitored, probably because the NPDES permit addresses it. Concentrations average 0.04 mg/ ℓ and are well below the 0.5 mg/ℓ limit in the permits. Barium and arsenic are less frequently monitored but appear to be in the range of 0.05 mg/ ℓ for barium to 0.005 mg/ℓ for arsenic. Both of these values are well below the discharge limits.

Table 3.21 Summary of average discharge and water quality data for uranium mines in Wyoming and a comparison with NPDES limits

Project	Mine Type	Discharge m³/min	Radioactivity Total U mg/ℓ	Radioactivity Ra-226 pCi/ℓ	TSS	Major and trace constituents, mg/ℓ SO₄	Zn	Fe	Ba	Cd	As
1	U	0.85	0.95	3.92	87.2		0.08	1.25			
2	U	6.57	0.41	2.28	2.7	234	0.02	0.02	0.05		
3	U	0.70	0.02	7.41	8.8		0.01				
4	S	1.89	1.30	10.66	5.0		0.01				
5	S	3.60	0.63	3.94	11.1		0.14				
6	S	5.68	0.02	2.85	10		0.05				
7	S	3.52	0.98	0.67	19.4	875	0.05			0.004	0.005
8	S	1.21	0.14	3.03	17.3		0.02				
9	S	0.10	1.14	3.6	62.5		0.16				
10	S	4.55									

All Mine Types (1 through 10):

Average:		2.87	0.62	4.26	24.9	555	0.06	0.64	0.05	0.004	0.005
Standard Deviation:		2.25	0.50	3.00	29.5	453	0.06	0.87			

Underground Mines (1 through 3):

Average		2.71	0.46	4.54	32.9	234	0.04	0.64	0.05		
Standard Deviation:		3.35	0.47	2.62	47.1		0.04	0.87			

Surface Mines (4 through 10):

Average		2.94	0.70	4.1	20.88	875	0.071			0.004	0.005
Standard Deviation:		1.96	0.53	3.4	21.04		0.063				

Summary of NPDES Permit Limits

Daily Average/Daily Maximum			2/4	3/10(a)	20/30		0.5/10	-/2	-/1	0.05/0.1	0.5/1
				10/30 Total Radium							

(a) Total Ra-226 limit is not monitored.

Source: NPDES permits from Region VIII (R. Walline, written communication), site-specific reports (EIS, ER), and self-monitoring data.

Mean values from six surface mining projects in Wyoming were the basis for estimating the effects of mine discharge on water quality. Values from mines in the South Powder River Basin model area compare very well with the Wyoming mines, thus supporting adoption of a model mine in the Basin. There were no strong differences in water quality between surface and underground mines. Table 3.21 shows that discharge is highly variable, ranging from 0.1 to 6.57 m^3/min, with an average of 2.87 m^3/min. In surface mining projects, the average is 2.94 m^3/min, with a standard deviation of 1.96, indicating considerable discharge variation among facilities. This study assumes an overall average flow of 3 m^3/min from each surface mine in the calculations of chemical loading of local and regional streams (see Section 3.3.3 and Appendix H).

Table 3.22 shows water quality and flow rates associated with open pit mines in other areas and in various stages of operation. Ongoing development of an open pit mine in Colorado involves 28 m^3/min discharge and is therefore well above the average. Radium, uranium, and suspended solids are relatively low. Producing open pit mines in New Mexico are usually dry or nearly so and are dewatered at rates of 0.6 m^3/min or less. The water is used for dust control. Radium concentrations can be very high (New Mexico Projects) due to long residence time of groundwater in the ore body and the concentrating effects of evaporation. Similary, groundwater associated with ore bodies in Texas and Wyoming may contain several hundred picocuries per liter.

Mine dewatering has the greatest potential for adverse environmental and public health impacts. Although contaminant concentrations in the effluent conform to NPDES requirements, there is long-term contaminant loading to the ambient environment. Contaminants concentrate on stream sediments because of sorption and evaporation and become available for transport by flood water. Regional or at least local dewatering of ore bodies may deplete high quality groundwater. Theoretically, dewatering may induce horizontal or vertical influx of poorer quality groundwater into productive or potentially productive aquifers, but the extent of this phenomenon is poorly documented. We strongly recommend further study because the work done to date is largely oriented toward determining engineering feasibility versus the overall environmental impact.

Table 3.22 Water quality associated with surface and underground mines in various stages of construction and operation

Project	Discharge m³/min	Total U mg/ℓ	Dissolved Ra-226 pCi/ℓ	Pb-210 pCi/ℓ	TSS	Milligrams per liter			
						SO$_4$	As	Mo	Se
Colorado									
Open pit mine:									
Development stage	28	1.044	4.10		16.2				
New Mexico									
Producing open pit mine, seepage to pit	0.13	2.5	180	17	168	2151	0.005	0.018	0.019
Open pit mine, ponded inflow water	0.58	2.6	220	26	23	842	0.005	0.545	0.043
Texas									
Active open pit mine holding pond.			50 to 100			380	< 0.01	< 0.01	< 0.01

Overland flow is not dismissed herein as a significant pathway, although its impact is of lesser importance according to data from April 1979 field studies in New Mexico and Wyoming (see Section 3.2.3.2 and Appendix G). A recent U.S. Geological Survey study for the Bureau of Indian Affairs (Ku79) addresses projected effects of runoff over long time periods if wastes and sub-ore are not stabilized or covered. The study concludes, with essentially no real data, that stream flows are too small in the sub-basin to transport wastes. In the larger basins, such as the Rio Puerco, sediment loads are so great that addition of tailings and, presumably, mine wastes would be insignificant. It is our opinion that additional field study is needed. Overland flow in a long time period could move radionuclides in the wastes into the main stream channels. Since this source will be available for many years after mine closure, if wastes are not stabilized, it may become a major one.

Seepage of contaminated water from mine holding ponds, which are operated to reduce suspended solids concentrations in mine discharge water, is believed to be insignificant. Since the ponds have relatively small areas, their seepage losses are small compared to losses by infiltration of releases to the watercourses. In some mining areas, such as the Powder River Basin, shallow groundwater quality is naturally poor. Maximum attenuation of contaminants is expected in the shallow, poorly permeable bedrock strata of the Wasatch and Fort Union Formations.

3.3.3   Hydraulic and Water Quality Effects of Surface Mine Discharge

3.3.3.1   Runoff and Flooding in the Model Surface Mine Area

3.3.3.1.1 Study Approach

Precipitation and runoff estimation for the model surface mine scenario in Wyoming considers three hydrographic units: sub-basin, basin, and regional basin. Respective surface areas are 11.4, 5,400, and 13,650 square kilometers (km^2). The mine is located in the sub-basin. The sub-basin, the basin, and the regional basin are all drained by ephemeral streams. The latter is drained by a major regional river that has wide seasonal variations in flow and is dry or nearly so about 180 days each year. The sub-basin has similar flow variability. Figure 3.9 depicts the mine in relation to the sub-basin, basin, and regional basin.
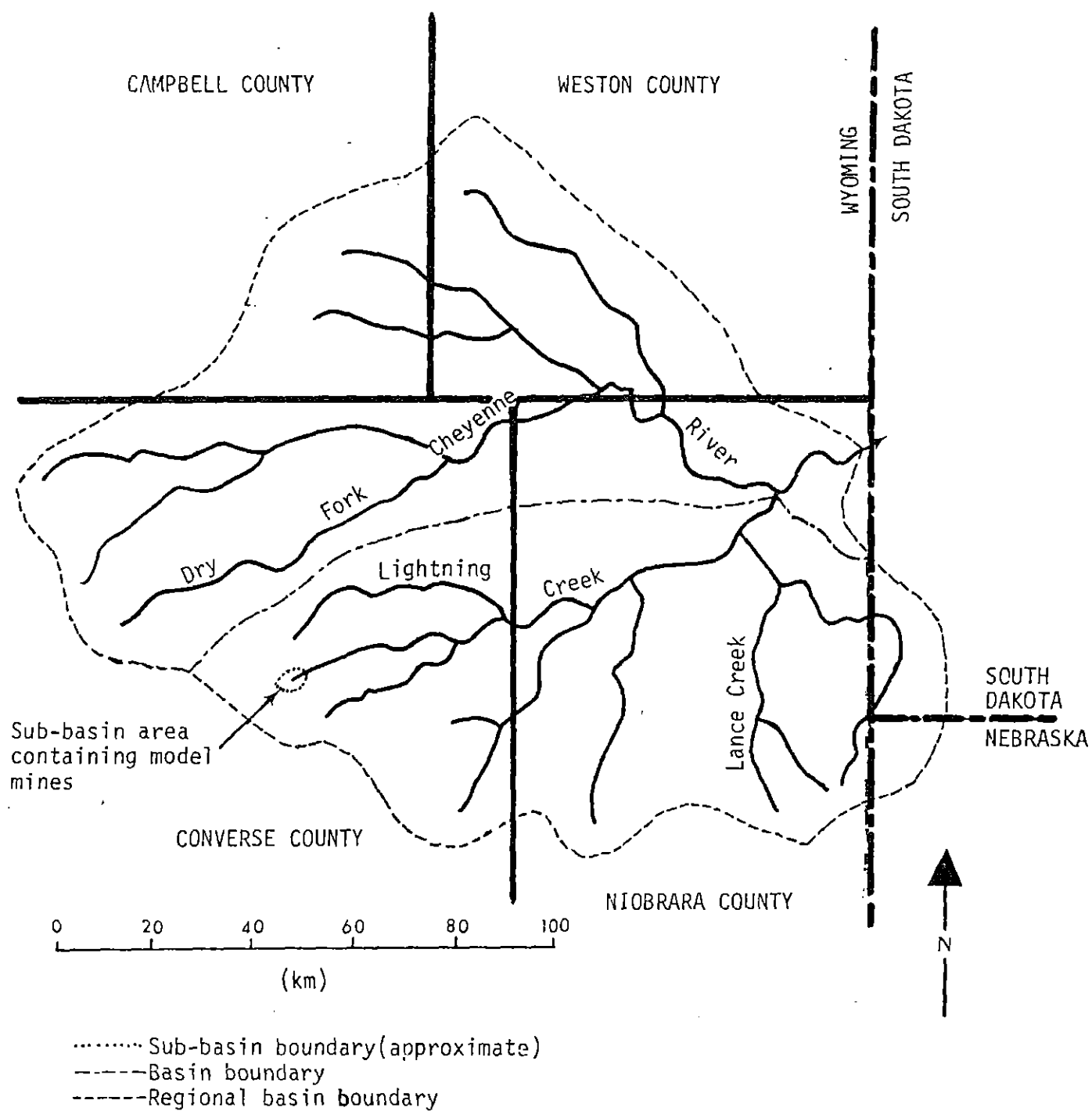
Figure 3.9 Sketch of sub-basin, basin, and regional basin showing orientation of principal drainage courses, areas of drainage, and location of mines.

The first general approach defined quality and volume of mine water discharge. Hypothetical hydrographic basins were then delineated and flood flows calculated for return periods ranging from 2 to 100 years. The individual and collective effects of discharge from three mines were then evaluated in terms of perennial flow, flood flow, and chemical transport. Of key importance was an estimation of the extent of perennial streams created by mine discharge and the influence of contaminants on water quality in the river draining the regional basin.

3.3.3.1.2 Description of Area

We selected an area of active mining and milling in the South Powder River Basin of Wyoming for analysis. The area has four active or imminently active uranium mills and a number of open pit mines. Available data on the geology, hydrology, and water quality of the area are sparse, but because of the mining and milling activity are relatively well known for a remote region like northeastern Wyoming. The study team chose one mining and milling project in the area for field investigation in April 1979; hence, additional data became available and are used herein as appropriate.

Terrain in the area has low rolling hills and an average elevation of 1414 m (MSL datum). Since the climate is not very different from that of nearby Casper, Wyoming, meteorological data from that station are fairly representative of the region. There are no relatively large seasonal and annual variations in precipitation intensity, frequency, and duration. Mean annual precipitation over a 30-year record period is 28.5 cm and occurs mainly as scattered thunderstorms in late spring and early summer. These thunderstorms supply 25 to 50 percent of the total annual precipitation and are usually of high intensity, short duration, and can be quite local. Potential pan evaporation averages 110 cm per year and greatly exceeds precipitation.

Streams in the study area are ephemeral and only exhibit measurable surface flow during snowmelt and heavy thunderstorm activity. Average total monthly flow for the period 1948 through 1970 for Lance Creek and the Cheyenne River at Spencer, Wyoming reveal distinct high- and low-flow periods in the year (Fig. 3.10). We believe that the streams represent the basin and regional basin hydrographic units used herein. Large watersheds usually exhibit measurable surface flow for about 180 days per year. Small watersheds, 30 to 40 square kilometers, may not flow at all for several consec-
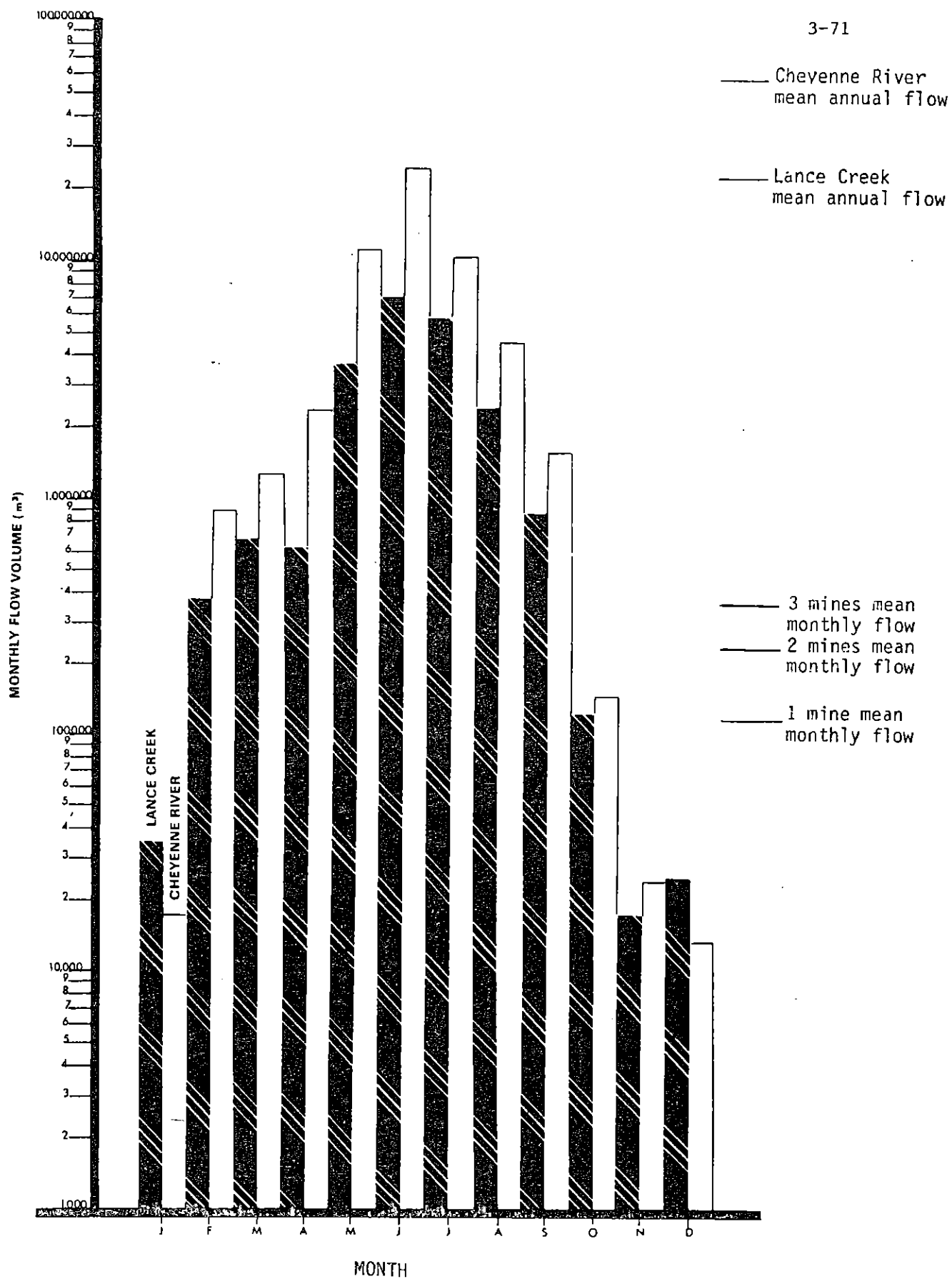
Figure 3.10 Average monthly flows for the Cheyenne River and Lance Creek near Spencer, Wyoming, for the period 1948-1970. (DOI59, DOI64, DOI69, DOI73)

utive years. Mean annual runoff is 0.8 to 1.3 cm or 0.0023 to 0.004 m^3/sec per km^2.

Peak flows in the regional basin and basin area are a result of snowmelt in at least 50 percent of the cases. This is commonly due to temporary but rapid melting from January to March. High flows can also result from widespread summer storms, but these are the exception. For small basins on the order of forty square kilometers or less, peak flows occur because of thunderstorms in the summer months. Thus peak flows in small basins versus the basin or regional basin commonly occur for different reasons and at different times in the year. A period of peak runoff from the sub-basin might coincide with a low flow or zero discharge condition in the basin or regional basin.

In the area of the Morton Ranch project (DOA75) there are 14 sub-basins, the average of which is about 11.4 km^2. Channel slopes are 11.4 to 31.4 m/km (average 21.2 m/km), and basin slopes are about 88 m/km. These are tributary to larger streams with channel slopes of 2.17 to 17.0 m/km (average 6.63 m/km), and which drain basins with an area of 5,400 km^2 and a mean annual flow of 0.80 m^3/sec. These, in turn, are tributary to a regional basin with an area of 13,650 km^2 and mean annual flow of 1.47 m^3/sec. All three hydrographic units are drained by ephemeral streams. The main stem of the regional system is dry an average of 180 days per year. The basin drains into the regional basin, assumed here to be the Cheyenne River Basin, which drains an area of 13,650 km^2 (Da75, Lo76, Ra77).

Surface water in the model area is used mainly for stock watering and irrigation. The amount of irrigated area in the basin is 1400 hectares, compared to 2800 in the regional basin. Because of extreme variability in surface flow volume and water quality, almost all municipal water comes from wells completed in bedrock. Stock water is from both wells and impoundments, whereas single-family domestic supplies are primarily from wells.

3.3.3.1.3 Method of Study

Because of dilution considerations, flow volume rather than peak discharge rate is of prime concern. For the basin and regional basin areas, only peak flow rate can be readily estimated on a probability basis for annual and longer time periods of perhaps 2, 5, 10, etc. years. Peak flow in the larger hydrographic areas commonly does not coincide with that in the

smaller basins. Also, there is poor correlation between peak flow rate (Q) and total flow volume (V) for streams draining large basins. Total flow volume in the larger basins can be estimated from partial duration flow data. That is, we can estimate the percentage of the time, during the year, flow will be of a given magnitude.

Relationships among runoff volume, rainfall, and surface area in small basins (encompassing less than 30 square kilometers) in the Powder River Basin have been developed by the U.S. Geological Survey (Cr78) and the Soil Conservation Service (DOA75). Peak discharge and total annual flow in the basin and regional basin units were measured by the U.S. Geological Survey for Lance Creek at Spencer, Wyoming and for the Cheyenne River near Spencer.

We analyzed the effects of perennial or chronic mine discharge on changing existing ephemeral streams in the sub-basin, basin, and regional basin to perennial streams using a crude seepage and evaporation model. The basic equations and approach, explained in Appendix H, are similar to those used in the Generic Environmental Impact Statement on Uranium Milling (NRC79b). Adjustments were made for mine discharge rates and infiltration and evaporation losses. The main output of the model is an estimate of which stream segments might become perennial and what the net discharge would be from a number of mines operating in the same sub-basin. Water quality impacts can only be very roughly assessed. For the time being, we assume that infiltration and evaporation decrease flow but do not effect the chemical mass in the system. That is, we assume contaminants in mine drainage are deposited on or in the stream/wash substrate and remain available for transport by flood water.

The sub-basin is as shown in Fig. H.1 (Appendix H) and contains three active uranium mines, each of which discharges 4,320 m^3/day. Quaternary alluvium constituting the channel is assumed to have a porosity of 40 percent. The sub-basin contains seven streams or wash segments, three receiving mine water directly. Water from the mines dissipates by infiltration, evaporation, and as surface flow that may leave the sub-basin entirely. Appendix H shows the basic equations and assumptions and gives a complete summary of "losses" due to seepage and evaporation as well as any net outflow from the sub-basin.

Precipitation-runoff in the Wyoming study area correlates rather closely to basin size. Basins of about 10,000 km^2 area have an annual unit-area runoff of 0.43 cm/yr; whereas an area of perhaps 25 km^2 might have a runoff of only 5 cm/yr. Decreased runoff (on a unit area basis) associates with larger basins and reflects water storage, channel losses, and evapotranspiration that occur mainly in the tributaries. Impoundments are rarely on the main stem of streams, where washouts are a problem, but rather on tributaries. The average impoundment is located about every 130 square kilometers, is rather small, and is used for stock water. Very infrequently, small flood-irrigation projects may use impounded water for grasslands. Seventy-five percent of the annual runoff occurs during the summer thunderstorm activity in May, June, and July. Snowmelt occurs rather slowly and is captured in the headwater areas, whereas rainfall events are rather intense and localized, causing excess flows that reach the main stem, Lance Creek and Cheyenne River. Sediment loads are high in both the tributary and main stem streams.

Contaminant concentrations in overland and channel flow during peak runoff events in the sub-basin are expected to follow the pattern shown in Fig. 3.11, the data for which are from the U.S. Geological Survey (H. Lowham, in preparation) for a small basin, Salt Wells Creek, in the Green River Basin of southwestern Wyoming.

Note in the inset of Fig. 3.11 that the washoff peak, that portion of the runoff enriched in dissolved and suspended materials, precedes the runoff peak. Runoff in small basins is typically associated with brief but intense thunderstorms that flush the land surface. Total suspended solids (TSS) concentrations are disproportionately high in the peak flow events. Discharges of 170 m^3/min carry 100,000 mg/ℓ TSS; whereas flows of 1 cfs might carry only 500 mg/ℓ. The leading edge of the high flow has the greatest concentration of suspended solids and dissolved chemical load. Figure 3.12 depicts discharge and specific conductance values as a function of time for the same small basin in Wyoming. Specific conductance (SC) is a rough measure of the total dissolved solids (DS) content, following the approximate relationship: DS = 0.71 SC. Note that the first rise in the flow hydrograph occurs about three hours after the peak for specific conductance, indicating the presence of a contaminated "front" laden with salts and other suspended and soluble materials. The second peak on the flow hydrograph similarly precedes and is associated with degraded water quality due to this flushing
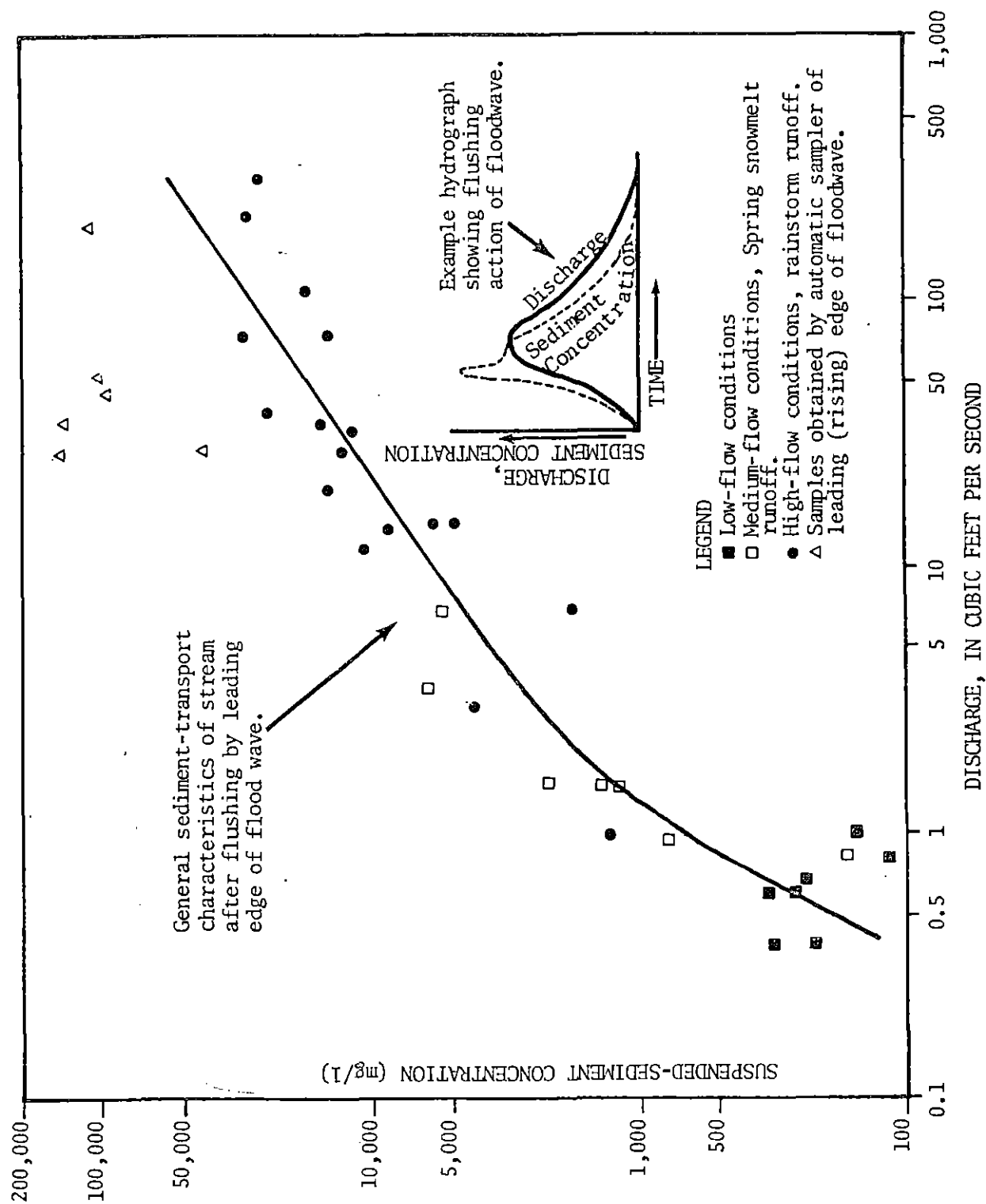
Figure 3.11 Suspended sediment concentration to discharge. Salt Wells Creek and tributaries, Wyoming (From U.S. Geological Survey data, H Lowham, in preparation)
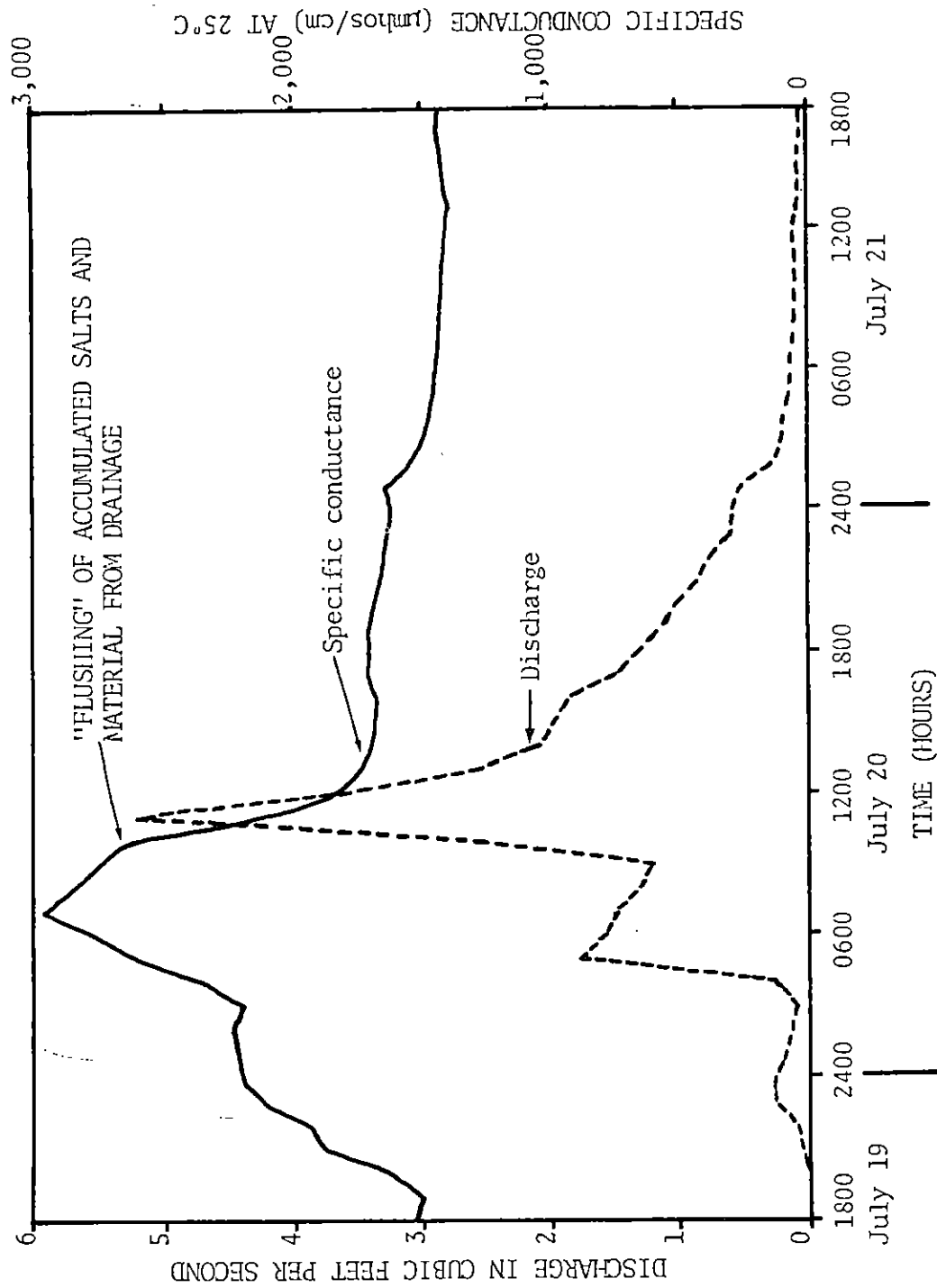
Figure 3.12 Relation of discharge and specific conductance to time at Salt Wells creek, Green River Basin, Wyoming. (From U.S. Geological Survey data, H. Lowham, in preparation.)

action. About 33 hours after precipitation begins, runoff water quality and flow very nearly approximate antecedent conditions. This indicates rather thorough flushing, most of which occurred in an 18-hour period.

Assuming similarity between the surface mining area and the situation described above, we believe intense flows of rather short duration flush most of the contaminants from the land surface and stream channels. Although sub-basin floods are expressed in terms of return period for Wyoming and in terms of partial duration (1-day, 7-days) and varying return periods for New Mexico, we believe the basic approaches (total flow vs. partial duration) to be rather similar because of the "flashy" nature of runoff in both study areas. In the New Mexico case, the mean discharge rate and the flow volume for the 7-day event are very often less than that for the 1-day event for the same return period. This also confirms the intense, short-term nature of runoff processes in the Wyoming and New Mexico model areas.

3.3.3.1.4 Discussion of Results

This section addresses the interaction between mine drainage and flood waters. Flood magnitude is addressed first, followed by calculation of water quality effects due to mine water. The U.S. Geological Survey technique (Cr78) for estimating floods in small basins in northeastern Wyoming was used to estimate peak discharge and total flow volume in the sub-basin. Multiple regression analysis reveals that the variables of area, slope, and relief provide roughly 90 percent correlation between rainfall and runoff (Cr78). Considering the numerous assumptions made throughout the analysis, only the area variable is used herein. It accounts for 70 percent of the flow. Table 3.23 shows the peak discharge rate and total flow volume from the sub-basin for floods with recurrence intervals (r) of 2 to 100 years. The basic equation for calculating discharge rate or flow volume is--

$$Q_r \text{ or } V_r = a \, A^{b_1} \tag{3.1}$$

where a = regression constant

b_1 = drainage area coefficient for area: peak discharge area: volume relationships

A = basin area

$= 11.4 \text{ km}^2$

Q_r, V_r = discharge rate and flow volume for flooding events with return periods of 2, 5, 10, 25, 50, 100 years.

The flow volume for the two-year flood is 32,921 m^3, and the instantaneous peak flow rate is 387 m^3/min. For comparison, we assume that the model mine discharges 3.00 m^3/min or 1.6 x 10^6 m^3 per year. Assuming the annual flood volume is 30,000 m^3, it is apparent that annual dilution is essentially nil and can be expected to be zero for perhaps 8 to 10 months of the year when there is no natural runoff. As Table 3.24 shows, flow volumes calculated using the Soil Conservation Service (DOA75) methodology average 1.5 times greater than those derived using the USGS approach. The latter are used herein in the interests of conservatism, i.e., there is less volume for dilution of contaminants.

Table 3.23   Peak discharge and total volume for floods of 2, 5, 10, 25, 50 and 100 year recurrence intervals

Recurrence Interval, r, in years	Regression Constant, a	b_1	Volume (m^3)	Regression Constant, a	b_1	Peak Discharge (m^3/min.)
2	9.62	0.689	32921	96.21	0.582	387
5	18.08	0.713	64116	199.6	0.612	840
10	24.87	0.727	90009	292.8	0.632	747
25	34.71	0.739	127862	441.1	0.660	1269
50	42.82	0.748	159920	575.4	0.679	2674
100	51.58	0.756	194937	731.1	0.699	3500

Peak flood discharges in the basin and regional basin range from 4,053 to 31,401 m^3/min for recurrence intervals of 25 years or less. For 50- and 100- year flooding events, peak discharge approximates 19,370 to 44,860 m^3/min. In the regional basin, discharge is 1.7 m^3/min or less approximately eight months of the year and equals or exceeds 17 m^3/min for about three months of the year, typically in the winter and early spring.

Maximum discharge from the basin and sub-basin is expected in the late spring and early summer months because of thunderstorms. At this time, flow in the river draining the regional basin is also at or near maximum, thus there is high probability for considerable dilution of runoff contaminated by mine drainage.

Total flow volumes for the basin and regional basin were estimated from U.S. Geological Survey records for the period 1948 to 1970. Figure 3.10 shows average monthly flows in cubic meters for the Cheyenne River and Lance Creek near Spencer, Wyoming. Immediately apparent is the close similarity in overall runoff pattern for the year.

Table 3.24   Summary of calculated total flow in the Wyoming model area sub-basin using the USGS and SCS methods

Recurrence Interval, r, in years	Sub-basin Total flow (m^3)[a]	Sub-basin Total flow (m^3)[b]
2	32,921	14,467
5	64,116	NC[c]
10	90,009	98,419
25	127,862	170,815
50	159,920	231,618
100	194,937	295,257

[a] Source: Cr78.

[b] Source: DOA75.

[c] NC = Not calculated.

Minimum flows occur in November, December, and January, and peak runoff in both basins occurs in May, June, and July. Long-term average annual flow in the basin is 2.18×10^7 m^3 and 5.64×10^7 m^3 in the Cheyenne River. These are almost exactly proportional to the respective basin areas of 5,360 km^2 and 13,650 km^2, indicating similar climatic and runoff conditions.

Assuming there are 3 mines operating for a 17-year period and that each mine discharges on the average 3.00 m^3/min continuously, total annual flow volume from the mines is 4.7 x 10^6 m^3. Cumulative discharge from the sub-basin is 7.04 m^3/min or 3.7 x 10^6 m^3/yr, which causes development of a perennial stream 12.8 km long within the basin. Insofar as the basin channel length is 141 km, the perennial stream ceases to flow well within the basin.

Appendix H explains the methodology and intermediate steps involved in deriving these foregoing values. Mine drainage water is not expected to flow the full length of Lance Creek or reach the Cheyenne River. However, on the basis of total monthly flow, the volume of mine drainage from one mine exceeds the flow in Lance Creek and the Cheyenne River for three months of the year, whereas flow from three mines exceeds basin flow for five months and regional basin flow for four months each year (Fig. 3.10).

The aqueous pathway for mine drainage is considered in terms of chronic, perennial transport in the mine water, per se, and transport by flood waters that periodically scour the channels where most of the sorbed contaminants would be located. Considering the random nature of flooding and the resulting uncertainty as to when the next 2-, 5-, or 10-year, etc. flood may occur, it is assumed that most contaminants accumulate on an annual basis and are redissolved by floods of varying return periods (2 to 10 years) and volumes. Many combinations of buildup and flooding are possible, such as buildup for 5 years or 10 years with perhaps several 2-year storms and one 5-year storm. Insofar as numerous assumptions are made in calculating volume and quality of mine discharge, basin runoff, and fate of the contaminants in the aqueous system, use of annual accretion and varying flood volumes in the sub-basin is considered adequate for estimating flood water quality.

Dilution of contaminated flows originating in the sub-basin and extending into the basin were conservatively calculated by assuming that the total flow during the low period equaled the mean annual flow. Thus, high flows and associated increased dilution are ignored, tending to make the analysis conservative. Contaminated flows from the sub-basin are diluted into these adjusted mean annual flows. Definition of the source term on an annual basis is most compatible with the radiation dose and health effects calculations in Section 6. Use of the low flow segment of the total annual

flow regime is decidedly conservative since total flow during the five months of low flow conditions amounts to 111,610 m^3 and 218,336 m^3 for the basin and regional basin, respectively. Average annual flow for the period of record (22 years) is considerably higher, amounting to 2.184 x 10^7 m^3 for the basin and 5.64 x 10^7 m^3 for the regional basin.

Runoff in the basin and regional basin is expected to markedly dilute contaminated flood flows originating in the basin. Such floods would scour contaminants from about 23 kilometers of channel affected by contaminants from the three active mines. Peak runoff events in the sub-basin are most likely in the late spring-early summer season when runoff in the basin and regional basin is the maximum or near maximum, on the average. However, peak runoff from the sub-basin could also occur when the basin and regional basin are at low flow or zero discharge. Such contrasts are present between the basin and regional basin flow regimes. From September through December, Lance Creek can be expected to have no discharge from 45 to 65 percent of the time, whereas the Cheyenne River will be dry, on the average, from 65 to 85 percent of the time (Fig. 3.13). Thus there is a distinct chance that contaminants transported in Lance Creek would not be immediately diluted upon reaching the Cheyenne River.

Before discussing the calculated concentrations of contaminants in the basin and regional basin streams, several other conditions need to be mentioned. In water-short regions like Wyoming, extensive use is made of impoundments to capture and store runoff. On Lance Creek, the model for the basin, the volume of existing impoundments is 15.78 x 10^6 m^3 or 72 percent of the annual average runoff. In the regional basin, modeled after the Cheyenne River, there are 4.2 x 10^7 m^3 of storage volume, which is 74 percent of the average flow of 56.4 x 10^6 m^3. Thus, it is very likely that discharge from the sub-basin or basin will not exit the basin, particularly in the periods of low flow. Contaminant concentrations, particularly those affected by sorption and precipitation reactions, are likely to be reduced as a result of sedimentation and long residence time in the impoundments, although there is some potential for overtopping, disturbance by cattle, and so on. Significant adverse impacts are not likely considering precipitation and sorption reactions which are likely to remove contaminants from the food chain. Proof of this is lacking and we recommend confirmatory studies for the stable elements. Previous studies (Ha78; Wh76) emphasized radiological contaminants.
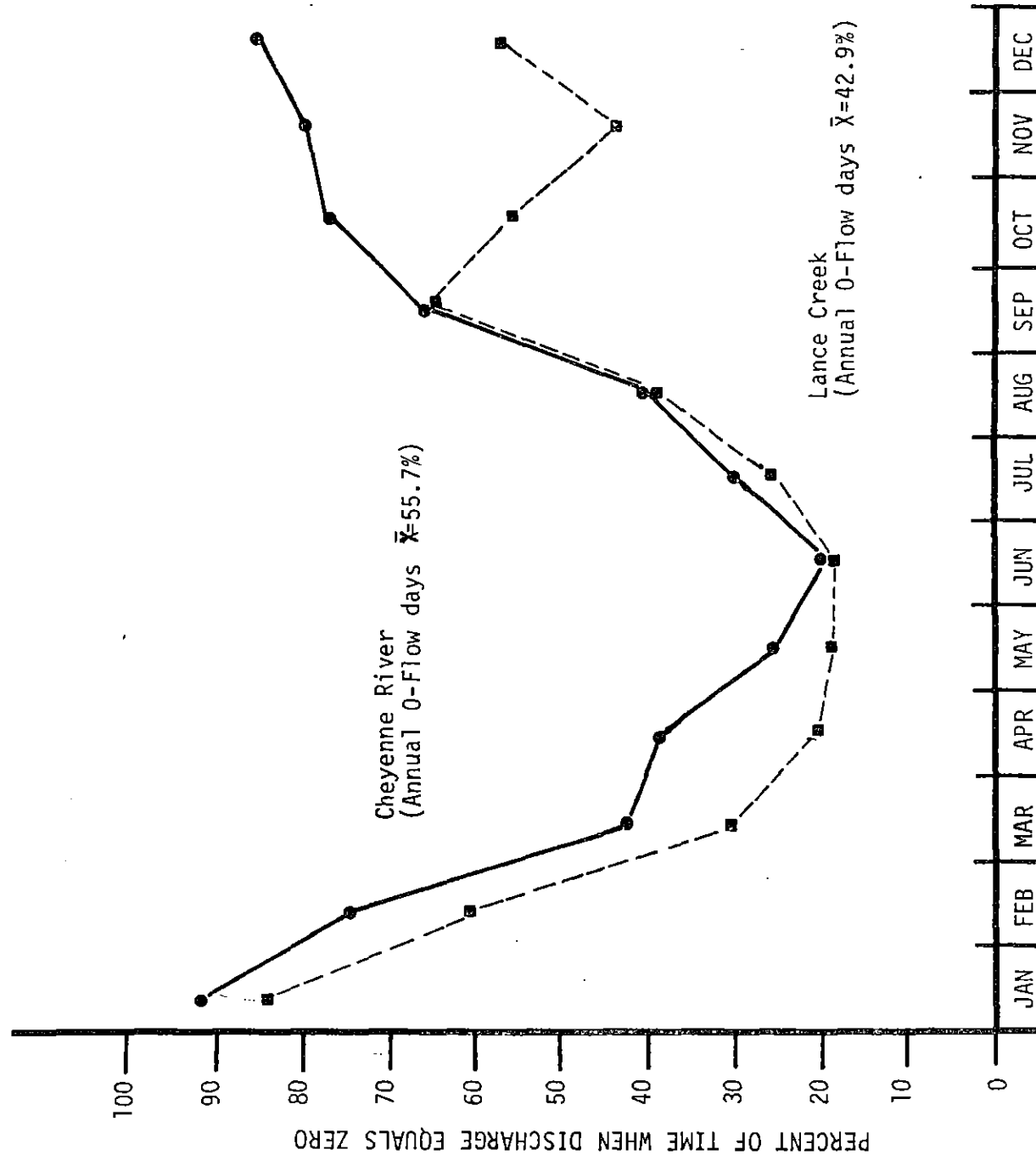
Figure 3.13 Periods of no flow in Lance Creek and the Cheyenne River near Riverton, Wyoming for the period 1948-1978 (Summarized from flow records provided by H. Lowham, U.S. Geological Survey, Cheyenne, WY.)

Radium-226 is strongly sorbed onto stream sediments and (or) it is subject to precipitation. Partial re-solution in subsequent floods occurs but it is assumed that only 10 percent of the mass deposited on an annual basis goes back into solution in flood waters. The rationale for this assumption is based on laboratory studies (Sh64; Ha68), field data from New Mexico (Ka75; Ku79), and review of the literature. Pertinent field and laboratory data specific to surface water quality in the Wyoming uranium mining areas are scarce, although studies by the State (summarized by Harp, 1978) are noteworthy. Sulfate is regarded herein as rather mobile and, as such, most of it infiltrates the shallow aquifer. Therefore, only 20 percent of the mass from a given mine on an annual basis is assumed available for re-solution in flood waters. The fate of zinc, arsenic, and cadmium is insufficiently understood to predict what fraction in the mine discharge will be removed from solution versus remain available for re-solution. Studies along these lines are necessary. Similarly, not all of the contaminants potentially present in mine waters from Wyoming are necessarily shown in Tables 3.21 and 3.25, which were developed based on available data from NPDES permits, environmental reports, and environmental impact statements. In the case of suspended solids, there is no calculation of non-point source contributions from mined lands. Sediment loads from such sources could be locally significant, but mined land reclamation and natural recovery seems to effectively mitigate problems. Only suspended solids from mine drainage, per se, are considered.

Table 3.25 shows the flood flow volumes (in the sub-basin) associated with events having return periods of 2, 5, 10, 25, 50, and 100 years. Also shown are the contaminant concentrations calculated from the annual contaminant loading diluted into the foregoing floods. As expected, concentrations are high because of the low dilution volumes associated with the small sub-basin. Surface water in the sub-basin might be impounded therein for use by stock or, less possibly, irrigation, but it is more likely that the principal impoundments would be in the larger hydrographic unit, the basin. The flood flow volumes shown represent runoff from the entire sub-basin. When the second and third mines begin to discharge, the annual loading and concentration values shown would have to be doubled or tripled. The reader should remember that background concentrations already present in flood runoff would be additive to the values in Table 3.25. However, these have been assumed

Table 3.25 Annual contaminant loading from one uranium mine and resulting concentrations in floods within the sub-basin for return periods of 2 to 100 years

Contaminant and concentration in mine effluent	Chemical mass available for transport on an annual basis	Flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 100 years[c]					
		$V_2 = 32921$ C_2	$V_5 = 64116$ C_5	$V_{10} = 90009$ C_{10}	$V_{25} = 127862$ C_{25}	$V_{50} = 159920$ C_{50}	$V_{100} = 194937$ C_{100}
Total uranium 0.070 mg/ℓ	110 kg/yr	3.34	1.72	1.22	0.86	0.69	0.56
Radium-226 4.1 pCi/ℓ	0.00065 Ci/yr[a]	19.7	10.1	7.2	5.1	4.1	3.3
Total suspended solids 20.9 mg/ℓ	32,955 kg/yr	1001	514	366	258	206	169
Sulfate 875 mg/ℓ	275,940 kg/yr[b]	8381	4304	3066	2158	1723	1416
Zinc 0.071 mg/ℓ	112.0 kg/yr	3.40	1.75	1.24	0.876	0.700	0.575
Cadmium 0.004 mg/ℓ	6.31 kg/yr	0.192	0.098	0.070	0.049	0.039	0.032
Arsenic <0.005 mg/ℓ	7.88 kg/yr	0.239	0.123	0.088	0.062	0.049	0.040

[a]Ten percent of the annual loading is assumed available for solution. The balance is assumed sorbed onto sediments or present in insoluble precipitates.

[b]Twenty percent of the annual loading is assumed available for transport and the balance is assumed to have infiltrated to the water table or it is present as an insoluble precipitate.

[c]V_r and C_r refer to, respectively, flood volume, in cubic meters, and concentration in milligrams per liter or picocuries per liter for an r-year flood. Concentrations are in milligrams per liter except radium-226, in pCi/ℓ.

Note.--Assumptions: Mine discharges continuously at a rate of 3.00 m^3/min and concentrations are the average of those shown in Table 3.21. All suspended and dissolved contaminants remain in or on the stream sediments and are mobilized by flood flow.

equal to zero in order to estimate incremental increases due to mining and to simplify the calculations.

Table 3.26 shows contaminant concentrations in the basin and regional basin streams from the discharge of one mine. For cases involving two or more mines, the concentration shown would be scaled up by a factor of two or more. Basically, the table shows the effects of taking contaminated flood waters from the sub-basin and diluting them in the low flow volume of the basin and regional basin. As expected, concentrations decrease with floods of greater volume and longer return period. Additional dilution occurs when discharge from the basin enters the regional basin. Taking the two-year runoff event in the sub-basin, for example, uranium is diluted from 3.34 mg/ℓ (Table 3.25) to 0.76 mg/ℓ in the basin and then to 0.44 mg/ℓ in the regional basin. There is some question as to whether the lesser sub-basin floods, particularly those with return periods of 25 years or less, would actually flow the length of the basin and enter the regional basin. Because much of the 22.7 km reach of stream directly affected by mine discharge is located in the basin, it is conservatively assumed that the contaminants will reach the basin and eventually the regional basin. The foregoing analysis is structured as a worst-case, maximum-concentration scenario.

Concentrations of contaminants in flood waters affected by mine drainage are compared to water standards for potable and irrigation uses (Table 3.27). Radium-226 concentrations in the basin and regional basin streams (Table 3.27) range from 1.6 to 4.5 pCi/ℓ and are below the drinking water standard (for Ra-226 + Ra-228) of 5 pCi/ℓ. Uranium concentrations range from 0.26 to 0.76 mg/ℓ, which is roughly equivalent to 176 to 514 pCi/ℓ. On the basis of chemical toxicity alone, such concentrations would probably present no problem for short periods, but radioactivity is another matter. Reevaluation of the standard for uranium in potable water is presently receiving attention within the Agency (R. Sullivan and J. Giedt, USEPA, oral communication, 1980). Briefly, there is consensus that the radiotoxicity of uranium is similar to that of radium-226 and 228. For continuous ingestion at a rate of 2 liters per day, it is suggested that potable water contain no more than 10 pCi/ℓ (0.015 mg/ℓ) natural uranium to reduce the incidence of fatal cancers to no more than 0.7 to 3 per year per million population (Office of Drinking Water guidance to the State of Colorado, July 7, 1979). Realizing that the

Table 3.26 Concentrations in basin and regional basin streams as a result of surface mine discharge

Parameter	Concentrations (mg/ℓ; pCi/ℓ in the case of radium) in basin discharge under low flow conditions due to influx of sub-basin floods with 2, 25, and 100 year return periods[a]			Concentrations (mg/ℓ; pCi/ℓ in the case of radium) in regional basin discharge under low-flow conditions due to influx of basin discharge, also under low-flow conditions, and sub-basin floods with 2, 25, and 100 year return periods[b]		
	C_2	C_{25}	C_{100}	C_2	C_{25}	C_{100}
Total Uranium	0.76	0.46	0.36	0.44	0.32	0.26
Radium-226	4.5	2.7	2.1	2.6	1.9	1.6
Total Susp. Solids	228	138	107	131	95	79
Sulfate	1909	1152	900	1098	797	668
Zinc	0.774	0.468	0.366	0.445	0.324	0.271
Cadmium	0.044	0.026	0.020	0.025	0.018	0.015
Arsenic	0.054	0.033	0.025	0.031	0.023	0.019

[a]Calculated as follows: Assuming a two year flood, uranium concentration in the outflow from the sub-basin equals 3.34 mg/ℓ and flow equals 32,921 m³ (see Table 3.25). Average total flow for 5 months of low flow conditions in the basin equals 111,610 m³. The concentration in the basin outflow, after dilution of the contaminated inflow from the sub-basin for floods of varying recurrence intervals equals:

$$C_{Basin} = \frac{V_{Sub\text{-}basin} \times C_{Sub\text{-}basin}}{(V_{Sub\text{-}basin} + V_{Basin})} = \frac{(32921 \text{ m}^3)(3.34 \text{ mg/}\ell)}{32921 \text{ m}^3 + 111610 \text{ m}^3} = 0.76 \text{ mg/}\ell$$

[b]Calculations similar to "a" above, except average total flow volume for 5 months of low flow in the regional basin equals 218,336 m³. Hence,

$$C_{Regional\ basin} = \frac{V_{Sub\text{-}basin} \times C_{Sub\text{-}basin}}{(V_{Sub\text{-}basin} + V_{Regional\ Basin})}$$

Table 3.27 Comparison of potable and irrigation water standards and surface water quality affected by surface mine drainage

Parameter	Range of contaminant concentrations in flood flow affected by mine discharge[a]				Potable water standards[b]		Irrigation[c]
	Basin		Regional Basin		Maximum Permissable Concentration (mg/ℓ)	Recommended Limiting Concentration (mg/ℓ)	Recommendations for maximum concentration for continuous use on all soils (mg/ℓ)
	Min.	Max.	Min.	Max.			
Total U	0.36	0.76	0.26	0.44	0.015/3.5/0.21[d]	---	---
Ra-226 + 228	2.1	4.5	1.6	2.6	---	5 pCi/ℓ	5 pCi/ℓ
TSS	107	228	79	131	---	---	---
Sulfate	900	1909	668	1098	---	250	200
Zinc	0.366	0.774	0.271	0.445	---	5.0	2.0
Cadmium	0.02	0.044	0.015	0.025	0.01	---	0.010
Arsenic	0.025	0.054	0.019	0.031	0.05	0.01	0.10

[a] Concentrations in milligrams per liter, except Ra-226 -228 which are in picocuries per liter.

[b] Sources: U.S. Environmental Protection Agency (EPA76) and, in the case of uranium, suggested guidance from the National Academy of Sciences (NAS79) to the USEPA and from USEPA (Office of Drinking Water) to the State of Colorado (La79).

[c] Source: NAS72.

[d] 0.015 mg/ℓ: Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis.

3.5 mg/ℓ: Suggested maximum 1-day limit based on chemical toxicity and intake of 2 liters in any one day.

0.21 mg/ℓ: Suggested maximum 7-day limit based on chemical toxicity and intake of 2 liters per day for 7 days.

limit of 10 pCi/ℓ (0.015 mg/ℓ) may not be cost effective, the Agency is contracting to develop the economic and technical basis for a uranium (in water) standard. The National Academy of Science, at the request of the Agency, evaluated the chemical toxicity of uranium. A maximum, 1-day concentration of 3.5 mg/ℓ (7 mg/day based on daily intake of 2 liters) is the "Suggested No Adverse Response Level" (SNARL). The corresponding concentration for a 7-day period is 0.21 mg/ℓ.

There are numerous complicating factors surrounding the foregoing suggested radiotoxicity and chemical toxicity limits for uranium. These include economic justification, technical feasibility, gut to blood transfer factors, and overall health of the receptor, to name a few. Of importance is the fact that a stricter standard for uranium in water is likely and that present NPDES limits of 1 mg/ℓ or previous drinking water limits on the order of 5 to 8 mg/ℓ are or will be superseded. For these reasons, the calculated uranium concentrations in the aqueous pathway are considered relative to the more recent, suggested limits of 0.015, 3.5, and 0.21 mg/ℓ.

Although mine effluents are not considered potable water, they infiltrate shallow aquifers that are potable in terms of the Safe Drinking Water Act. The extent to which shallow aquifers in uranium mining areas are used for potable water supply is presently small, but accurate surveys of well locations and water quality are scarce. If a limit of 0.015 mg/ℓ for uranium in potable water is set, it appears that uranium instead of radium-226 may be the primary pollutant of concern in both surface runoff and related shallow groundwater.

Of the remaining contaminants, sulfate and possibly cadmium might exceed drinking water standards. Cadmium may also limit use of the water for irrigation. These results provide only a rough estimate of water quality effects. There are other stable toxic elements to consider, but there are insufficient data. Multiple mine sources would increase the concentration, but ion exchange, sorption, etc. would reduce them. The net effect is simply unknown. It does appear that uranium, in particular, deserves additional study in light of new interpretations concerning radiotoxicity.

3.3.3.2 Impacts of Seepage on Groundwater

The previous analysis assumed no infiltration (to groundwater) of dissolved or suspended contaminants, thereby creating a maximum or worst-case situation with respect to transport via floodwaters. In fact, contaminants will also infiltrate through the stream deposits. Anions and selected stable elements like uranium, selenium, and molybdenum are most likely to migrate downward. Insofar as the alluvial, valley fill aquifer may be used locally, particularly in the case of larger drainage basins and the regional basin, some analysis of potential impacts is offered herein.

Effects of mine drainage impoundments used to settle suspended solids are excluded from the present analysis. Such impoundments are relatively small, commonly less than 1 or 2 hectares, and tend to become self-sealing due to settling of fines. Potable water supplies at the mines are usually from deep exploration borings converted to water wells or from mine water. Problems may exist with such water being contaminated, as has been documented in the Grants Mineral Belt (EPA75), but we do not believe seepage from settling ponds to be a factor.

Infiltration of water discharged to ephemeral stream courses was not calculated separately. It was combined into a lumped term incorporating infiltration and evaporation. Both losses are, in part, a function of surface area. Infiltration takes place primarily in the basin. When three mines are operating, 22.7 km of perennial stream is created and extends into a portion of the basin. Infiltration of the mine effluent adds primarily to the amount of water in storage in the alluvium, versus acting as a source of recharge to the deeper, consolidated strata.

As with many of the intermontane basins in Wyoming, water in the South Powder River Basin is primarily groundwater recharged by sporadic runoff from limited precipitation (Ke77). Some stock ponds that collect surface runoff are supplemented by groundwater from wells or springs. Mine water discharged from one underground mine is used to irrigate approximately 65 hectares of native grass, alfalfa, oats, and barley. In general, groundwater is not used for irrigation (Ho73). Groundwater use for domestic supplies is largely confined to the Dry Fork of the Cheyenne River (Ke77). The number of wells is close to a density of one per 400 ha (Ke77). Typical wells are completed in the alluvium and yield less than 100 ℓ/min.

Geological formations in the southern portion of the Powder River Basin include in descending order and increasing age; the 1) Alluvium, 2) Wasatch Formation, 3) Fort Union Formation, 4) Lance Formation, 5) Fox Hills Formation, and 6) older rocks too deep to be affected by uranium mining (NRC78c). Table 3.28 shows the well depth for each formation, anticipated well yields, and the total dissolved solids content in the vicinity of an active uranium mining and milling project in the South Powder River Basin.

Water quality in the Wasatch and Fort Union Formations ranges widely and appears to correlate with the permeability of the water-bearing sand and proximity to outcrops. No relation of water quality to depth is apparent. Analyses of water from Cenozoic rocks show dissolved solids ranging from less than 100 to more than 8000 mg/ℓ (Ho73). Of the 258 analyses performed by the USGS, 55 showed dissolved solids less than 500 mg/ℓ, 133 less than 1000 mg/ℓ, and 125 more than 1000 mg/ℓ. Sodium, sulfate, and bicarbonate are the dominant ions, and water is usually excessively hard. Iron is characteristically a problem in water from the Wasatch and Fort Union Formations (Ho73). Element distributions show considerable variability due to clay lenses in the sandy units (NRC78c). The clays act as barriers to groundwater movement and preferentially concentrate some elements. Table 3.29 shows the ambient groundwater quality in the immediate area of three active mills in the South Powder River Basin.

In the Wyoming model mine sub-basin, total inflow equals 9 m^3/min or 4.73×10^6 m^3/yr, and total annual infiltration loss equals 4.65×10^6 m^3 (calculated in Appendix H). Restated, 98.2 percent of the discharge infiltrates and the remainder evaporates.

Infiltration of 4.65×10^6 m^3/yr is not likely to continue for the full duration of mining unless the bedrock strata have the same or similar permeability as the alluvium and (or) there is an extensive zone of unsaturated alluvium to provide storage. The alluvium in the Wyoming study area is concentrated along the stream axes, is relatively thin, and is underlain by less permeable bedrock strata. It is probable that a zone of saturated alluvium will gradually develop and extend downstream as mine discharge continues. Recharge from the alluvium to the underlying Wasatch or Fort Union Formations will occur but at a low rate compared to infiltration. Water quality in the alluvium is highly variable (Table 3.29); it may or may not be affected by mine drainage. Adverse impacts, if any, are likely to be a result of uranium, sulfate, and mobile elements.

Table 3.28 Northeastern Wyoming groundwater sources

Geologic Period	Aquifer	Depth Range of Wells, m	Anticipated Well Yield, ℓpm		Total Dissolved Solids, mg/ℓ
			Common	High	
Quaternary	Alluvium	3-30	20-945	1140-2270	106-7340
Tertiary	Wasatch	12-300	4-150	380-2370	160-6620
	Fort Union	45-180	4-110	380	484-3250
Cretaceous	Lance	45-365	4-190	1900	450-3060
	Fox Hills	210-700	75-260	760-1900	1240-3290
	Mesaverde	12-915	57-150	225-265	550-1360
	Cody	30-335	4-20	380-760	6392-12,380
	Frontier	20-610	4-20	380-1135	390-2360
	Dakota	75-1830	95-380	760-3410	218-1820
Jurassic	Sundance	120-210	4-20	95	894-2310
Triassic	Spearfish	6-275+	4-115	380-760	2590
Pennsylvanian	Minnelusa	75-1980	95-950	1860-7470	255-3620
Mississippian	Pahasapa	150-2320	380-9460	26,500-35,600	290-3290
Ordovician	Bighorn	0-60	3785	3785	427-3219
Cambrian	Flathead	20-1800	760		124

Source: NRC78b.

Table 3.29 Groundwater quality of wells sampled by the three major
uranium producers in the South Powder River Basin, Wyoming

Parameter	Range of Concentration Reported (mg/ℓ)		
	Kerr-McGee[a]	TVA[b]	Exxon[c]
pH	7.4-8.0	7.4 - 8.5	7.3-8.1
Spec. cond.			
μmhos/cm	210-1100	250-1300	290-600
Ca	28-343	10-200	26-150
Mg	8-81	2-80	1-13
Na	5-71	10-300	54-121
HCO_3	30-380	70-110	90-412
SO_4	28-980	8-1000	58-575
Cl	<5-57	11-25	6-16
Zn	0.006-18.0	0.03 -3	ND- 0.14[d]
Fe		0.2 -20	0.01- 1.64
Ba			ND- 0.05
Radium (pCi/ℓ)	0.41 - 5.18	0.2 -18	0.4 -12.0
Uranium (mg/ℓ)	< 0.002- 2.3	0.002-60	0.0004 - 0.21

[a]Shallow wells up to 61 meters depth, Tables 2.6-7 through 2.6-10 of
reference Ke77.

[b]From Figs. C1 and C3 of reference NRC78b.

[c]Table 2.12 of reference NRC78d.

[d]ND: Not detectable.

An actual example of this saturated front developing and moving down-gradient is present at the Kerr-McGee Nuclear Corporation's Bill Smith Mine in South Powder River Basin (Ke77). The mine discharges to a tributary of Sage Creek at a rate of about 1.7 m^3/min. From the period January 1974 to late 1976, a flow front 23 km long developed as a result of infiltration into the sandy alluvium. The discharge water maintains a high groundwater level in the stream bed. Unfortunately, no information is available on the geometry of the stream channel to evaluate the volume of water that has infiltrated in the three-year period or on any water quality changes that have occurred.

In summary, additional field data are needed to properly address the water quality effects of infiltration. Both theory and at least one field example indicate extensive infiltration of effluent containing at least some mobile stable and radioactive contaminants. Therefore, we recommend additional field investigations to determine, at the minimum, any hydraulic and water quality effects of mine discharge on shallow aquifers and the influence of dewatering on regional water levels and water quality, regardless of pre-existing or anticipated local water use patterns.

3.3.4   Gases and Dusts from Mining Activities

Dusts and toxic gases are generated from routine mining operations. Combustion products are produced by large diesel and gasoline-powered equipment in the mine and by trucks transporting the overburden, ore, and sub-ore from the pit to storage pile areas. Dusts are produced by blasting, breaking, loading, and unloading rock and ore and by haulage trucks moving along dirt roads. Finally, Rn-222 will emanate from exposed ore in the pit and from the ore as it is broken, loaded, and unloaded. These sources will be discussed individually.

3.3.4.1 Dusts and Fumes

Most vehicular emissions are from the combustion of hydrocarbon fuels in heavy-duty, diesel-powered mining equipment. Surface mines produce considerably more emissions than underground mines, since the overburden must be removed before the ore can be mined. The principal emissions are particulates, sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. The quantity of these combustion products released to the atmosphere depends on the number, size, and types of equipment used.

The EPA estimates the following emissions from mining 1350 MT of ore per day from a surface mine (Re76).

Pollutant	Emissions per Operating Day, kg/d	
	Mining Operations	Overburden Removal
Particulates	17.0	18.9
Sulfur oxides	35.4	39.3
Carbon monoxide	294.2	327.4
Nitrogen oxides	484.6	538.4
Hydrocarbons	48.4	53.8

Assuming a 330 operating-day-year (Ni79), we adjusted these emission rates to ore production for the average surface mine (1.2×10^5 MT/yr) and the average large surface mine (5.1×10^5 MT/yr) as described in Sections 1.3.1 and 3.3.1. Table 3.30 shows the total airborne combustion product emissions. These estimated emission rates are somewhat higher than rates previously suggested by the U.S. Atomic Energy Commission (AEC74).

Table 3.30 Estimated air pollutant emissions from heavy-duty equipment at surface mines

Pollutant	Emissions, MT/yr[a]	
	Average Mine[b]	Average Large Mine[c]
Particulates	3	14
Sulfur oxides	7	28
Carbon monoxide	55	235
Nitrogen oxides	91	387
Hydrocarbons	9	39

[a] Based on (Re76) and 330 operating days per year (Ni79).
[b] Ore production = 1.2×10^5 MT/yr.
[c] Ore production = 5.1×10^5 MT/yr.

Dust is produced from blasting, scraping, loading, transporting, and dumping ore, sub-ore, and overburden. Additional dust is produced when the ore is reloaded from the stockpile for transportation to the mill. Dust emissions vary widely, depending upon moisture content, amount of fines, number and types of equipment operating, and climatic conditions. Because ore is usually wet, the relative amounts of dust produced from mining and handling it are usually small. We selected the following emission factors from those suggested by the EPA for the above listed mining activities (Hu76, Ra78, Da79):

Blasting = 5×10^{-4} kg dust/MT

Scraping and bulldozing = 8.5×10^{-3} kg dust/MT

Truck loading = 2.5×10^{-2} kg dust/MT

Truck dumping = 2×10^{-2} kg dust/MT

We applied these emission factors to the ore, sub-ore, and overburden production rates of the average mine and average large mine and estimated average annual dust emissions for these mining activities (see Table 3.31). These are probably maximum emission rates because blasting is not always required, and some emission factors appear to have been based upon data from crushed rock operations, which would contain more fines than rock removed from surface mines. One-half the emission factor values were applied to ore and sub-ore because they are usually wet, except when reloading ore from the stockpile, in which case it is assumed to have dried during the 41-day residence period (Section 3.3.1.2).

The movement of heavy-duty haul trucks is probably the largest single source of dust emissions at surface mines. An emission factor (EF) for this source can be computed by the following equation (EPA77b).

$$EF = 2.28 \times 10^{-4} \ (s) \ \left(\frac{V}{48}\right) \ \frac{365-W}{365} \ (TF) \ (f) \qquad (3.2)$$

where,

EF = Emission factor, MT/vehicle kilometer traveled (MT/VKmt),

S = Silt content of road surface, percent,

V = Vehicle velocity, kmph [Note: This term becomes $\left(\dfrac{V}{48}\right)^2$
   for velocities less than 48 km/hr (EPA77b, DA79)],

W = Mean annual number of days with 0.254 mm or more rainfall,

TF = Wheel correction factor, and

f = Average fraction of emitted particles in the <30 µm diameter sus-
   pended particle size range; particles having diameters greater
   than 30 µm will settle rapidly near the roadway.

Values selected for these terms in the solution of Equation 3.2 are --

S = 10 percent (Da79),

V = 32 km/hr for heavy-duty vehicles and 48 km/hr for light vehicles
   (therefore, the velocity term is $(32/48)^2$ and $(48/48)$, respectively),

W = 90 days (EPA77b),

TF = 2.5 (Da79) (heavy-duty vehicles only), and

f = 0.60, since the weight percent of particles of less than 30 µm
   and greater than 30 µm in diameter is generally considered to
   be 60 and 40 percent, respectively (EPA77b).

Substituting these values into Equation 3.2 yields 1.15×10^{-3} MT/VKmT and
1.03×10^{-3} MT/VKmt for the emission factors of heavy-duty haul trucks and
light duty vehicles, respectively.

Table 3.31 shows estimated dust emissions for the movement of heavy-duty
haul trucks using the following information:

Table 3.31 Average annual dust emissions from mining activities

| Mining Activity | Dust Emissions, MT/yr | | | | | |
| | Average Mine[a] | | | Average Large Mine[b] | | |
	Ore[c]	Sub-ore[c]	Overburden	Ore[c]	Sub-ore[c]	Overburden
Blasting	0.03	0.03	3.0	0.13	0.13	20
Scraping/bulldozing	NA[d]	NA	51	NA	NA	340
Truck Loading	1.5	1.5	150	6.4	6.4	1000
Total at Pit Site	1.53	1.53	204	6.53	6.53	1360
Truck Dumping	1.2	1.2	120	5.1	5.1	800
Reloading stockpiled ore[e]	3.0	NA	NA	13	NA	NA
Total at Pile Sites	4.2	1.2	120	18.1	5.1	800
Vehicular dust[f]	14	14	304	59	59	2020
Wind suspended dust from storage piles	10	3	30	44	10	94

[a] Based on annual production rates of 1.2×10^5 MT of ore and sub-ore and 6.0×10^6 MT of overburden.
[b] Based on annual production rates of 5.1×10^5 MT of ore and sub-ore and 4.0×10^7 MT of overburden.
[c] Assumed wet.
[d] NA - not applicable.
[e] Assumed dry.
[f] Dust emissions from heavy-duty vehicular traffic along ore, sub-ore and overburden haul roads.

$EF = 1.15 \times 10^{-3}$ MT/VKmt,

Truck capacities = 31.8 MT for ore and sub-ore and

109.1 MT for overburden (Da79),

Round-trip haul distance = 3.2 km to ore and sub-ore piles

and 4.8 km to overburden dump, and

Annual production rates = given in Section 3.3.1 and in the

footnotes of Table 3.31.

Additional dust emissions will occur from the movement of light-duty vehicles along access roads. Using the emission factor derived above (1.03 x 10^{-3} MT/VKmT) and assuming that there are 24 km of access roads traveled 4 times a day for 330 operating days per year, about 33 MT of dust will be produced from this source annually. Emissions during haulage road maintenance is relatively small and will not be considered.

Table 3.31 also shows average annual dust emissions from wind erosion of overburden, sub-ore, and ore piles at the model surface mines. For these computations, we assumed the model overburden pile to be that of Case 2 and in the shape of a 65-m high truncated cone (Table 3.11). The same was assumed for the average mine, except the pile height was 30 m. The sub-ore piles of both mines were assumed to have a truncated cone configuration (Table 3.20). The same configuration was also assumed for the ore piles, but the pile heights were 9.2 m for the average large mine and 3.1 m for the average mine (Table 3.17).

Emission factors, computed in Appendix I, are 0.850 MT/hectare-yr for overburden and sub-ore piles and 0.086 kg/MT for the ore stockpiles. The first emission factor was multiplied by the overburden and average sub-ore pile areas; the second factor was multiplied by the annual ore production.

In computing the Table 3.31 dust emissions, we assumed no effective dust control program and that there was no vegetation on overburden and sub-ore piles. Haul roads are normally sprinkled routinely during dry periods, and stabilizing chemicals are applied primarily to ore haul roadways at some mines. Sprinkling can reduce dust emissions along haul roads by 50 percent, and up to 85 percent by applying stabilizing chemicals (EPA77b, Da79).

The dust emissions from vehicular traffic (Table 3.31) (transportation) were summed with those produced by light vehicular traffic (33 MT/yr) and considered as one source of emissions. Concentrations of contaminants in the dust are unknown. Some spillage of ore and sub-ore along haul roads will undoubtedly raise uranium levels in roadbed dust. As an estimate, uranium and daughter concentrations in the dust were considered to be twice back-ground, 8 ppm (2.7 pCi/g), while concentrations of all other contaminants were considered to be similar to those in overburden rock (Section 3.3.1.1, Table 3.16). Table 3.32 shows the annual emissions computed with these assumptions.

Table 3.33 lists annual contaminant emissions from mining activities (scraping, loading, dumping, etc.) according to source location, at the pit and at the piles. Contaminant emissions were computed by multiplying the total annual dust emissions at each pile (Table 3.31) by the respective contaminant concentrations in each source -- overburden (Section 3.3.1.1.; Table 3.16), sub-ore (Section 3.3.1.3; Table 3.19) and ore (Section 3.3.1.2; Table 3.19). Contaminant emissions at the site of the pit were computed by multiplying the total annual dust emissions of ore, sub-ore, and overburden (Table 3.31) by their respective contaminant concentrations. The three pro-ducts of the multiplication were then summed to give the values in the 4th and 8th data columns of Table 3.33. The health impact of the sources at each location will be assessed separately in Section 6.1.

Table 3.34 lists annual contaminant emissions due to wind suspension and transport of dust. These values were computed by multiplying the annual mass emissions (Table 3.31) by the contaminant concentrations in overburden, sub-ore, and ore listed in Sections 3.3.1.1, 3.3.1.3, and 3.3.1.2, respec-tively. The uranium and uranium daughter concentrations were also multiplied by an activity ratio (dust/source) of 2.5 (Section 3.3.1.2). Although some metals may also be present as secondary deposits, it was believed that there were insufficient data to justify multiplying their concentrations by the 2.5 ratio.

3.3.4.2   Radon-222 from the Pit, Storage Piles, and Ore Handling

Rn-222 will be released from the following sources during surface mining operations:

Table 3.32 Average annual emissions of radionuclides (μCi) and stable elements (Kg) from vehicular dust at the model surface mines

Contaminant	Average Large Surface Mine[a]	Average Surface Mine[b]
Arsenic	20	3.3
Barium	630	106
Copper	39	6.6
Chromium	< 111	< 19
Iron	13,030	2,190
Mercury	< 17	< 2.9
Potassium	15,200	2,560
Manganese	1,050	177
Molybdenum	5.4	0.9
Lead	48	8.0
Selenium	4.3	0.7
Strontium	330	55
Vanadium	220	37
Zinc	43	7.3
Uranium-238 and each daughter	5,860	990
Thorium-232 and each daughter	2,170	370

[a]Mass emissions = 2,170 MT/yr.
[b]Mass emissions = 365 MT/yr.

Table 3.33   Average annual emissions of radionuclides (µCi) and stable elements (kg) from mining activities at the model surface mines

Contaminant	Average Surface Mine[a]				Average Large Surface Mine[a]			
	Overburden Pile Site	Sub-ore Pile Site	Ore Pile Site	Pit Site	Overburden Pile Site	Sub-Ore Pile Site	Ore Pile Site	Pit Site
Arsenic	1.1	0.10	0.36	2.1	7.2	0.44	1.6	13
Barium	35	1.1	3.9	62	232	4.7	17	406
Cobalt	NR[b]	0.02	0.07	0.05	NR	0.08	0.29	0.21
Copper	2.2	0.07	0.26	3.9	14	0.31	1.1	25
Chromium	<6	0.02	0.08	<10	<41	0.10	0.36	<70
Iron	720	19	66	1,270	4,800	80	284	8,360
Lead	2.6	0.09	0.33	4.7	18	0.40	1.4	31
Magnesium	NR	4.2	15	11	NR	18	63	46
Manganese	58	1.2	4.0	102	388	4.9	17	672
Mercury	<1	ND[c]	ND	<1.6	<6.4	ND	ND	<11
Molybdenum	0.3	0.14	0.48	0.86	2.0	0.59	2.1	4.9
Nickel	NR	0.02	0.08	0.06	NR	0.10	0.36	0.26
Potassium	840	30	105	1,500	5,600	128	453	9,850
Selenium	0.2	0.13	0.46	0.74	1.6	0.56	2.0	4.2
Strontium	18	0.16	0.55	31	120	0.66	2.4	206
Vanadium	12	1.7	5.9	25	80	7.2	26	154
Zinc	2.4	0.04	0.12	4.2	16	0.15	0.52	28
Uranium-238 & each daughter	1,800	120	2,990	4,300	12,000	510	12,900	25,700
Thorium-232 & each daughter	120	2.4	42	220	800	10	180	1,440

[a] Mass emissions from Table 3.31.
[b] NR - Not reported.
[c] ND - Not detected.

Table 3.34 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model surface mines

Contaminant	Average Large Surface Mine			Average Surface Mine		
	Overburden Pile	Sub-Ore Pile	Ore Stockpile	Overburden Pile	Sub-Ore Pile	Ore Stockpile
Arsenic	0.85	0.86	3.8	0.27	0.26	0.86
Barium	27	9.2	40	8.7	2.8	9.2
Cobalt	NR(a)	0.16	0.70	NR	0.05	0.16
Copper	1.7	0.61	2.7	0.54	0.18	0.61
Chromium	<4.8	0.20	0.88	<1.5	0.06	0.20
Iron	564	157	690	180	47	157
Mercury	<0.75	ND(b)	ND	<0.24	ND	ND
Potassium	660	250	1,100	210	75	250
Magnesium	NR	35	154	NR	11	35
Manganese	46	9.6	42	15	2.9	9.6
Molybdenum	0.24	1.2	5.0	0.08	0.35	1.2
Nickel	NR	0.20	0.88	NR	0.06	0.20
Lead	2.1	0.78	3.4	0.66	0.23	0.78
Selenium	0.19	1.1	4.8	0.06	0.33	1.1
Strontium	14	1.3	5.7	4.5	0.39	1.3
Vanadium	9.4	14	62	3.0	4.2	14
Zinc	1.9	0.29	1.3	0.60	0.09	0.29
Uranium-238 & each daughter	1,410	1,000	31,300	450	300	7,100
Thorium-232 & each daughter	94	20	440	30	6.0	100

(a)NR - Not reported.
(b)ND - Not detected.

1. Ore, sub-ore, and overburden during rock breakage and loading in the pit and unloading on the respective piles. (Since rock breakage, loading, transporting, and unloading usually occur in a short time period, they are considered one release.)
2. Ore during reloading from the stockpile after a 41-day residence time (Section 3.3.1.2).
3. Exposed surfaces of overburden, ore, and sub-ore in the active pit area.
4. Overburden, ore, and sub-ore pile surfaces.

The annual quantities of Rn-222 released from sources 1 and 2 above were computed using the following factors and assumptions:

1. Rn-222 is in secular equilibrium with U-238.
2. The density of ore, sub-ore, and overburden is 2.0 MT/m^3.
3. Annual production rates of ore, sub-ore, and overburden are those given previously in this Section and in footnotes "a" and "b" of Table 3.31.
4. All Rn-222 present, 0.00565 Ci/m^3 per percent U_3O_8, is available with an emanation coefficient of 0.27. [Although an emanation coefficient of 0.2 is commonly used (Ni79), recent emanation-coefficient measurements for 950 samples of domestic uranium ores by the Bureau of Mines indicate a value between 0.25 and 0.3 to be more appropriate (Au78, Tanner, A.B., Department of Interior, Geological Survey, Reston, VA, 11/79, personal communication). Therefore, an emanation coefficient of 0.27 was selected.]
5. The quantities of U_3O_8 present in ore, sub-ore, and overburden are 0.10, 0.015, and 0.0020 percent, respectively.

Substituting these values into the following equation yields the Rn-222 releases given in Table 3.35 for the average mine and the average large mine.

$$\text{Rn-222 (Ci/yr)} = (\text{Percent } U_3O_8) \left(\frac{0.00565 \text{ Ci}}{m^3 \times \text{percent}}\right) (0.27) \left(\frac{M^3}{2.0 \text{ MT}}\right) \quad (3.3)$$

$$\times \left(\text{Production Rate, } \frac{MT}{yr}\right)$$

The quantities of Rn-222 that emanate from exposed overburden, ore, and sub-ore surfaces in the pit were estimated by the following method. Exposed

surface areas of ore and sub-ore are assumed equal since equal quantities of each are mined. The computation assumes an ore plus sub-ore zone 12 m thick (h_1) in the shape of a truncated cone with 45 degree sloping sides (Fig. 3.14). The radii of the zone, r_1 and r_2, can be computed using the following equation from the relationship $r_2 = r_1 + 12$ and the volumes of ore plus sub-ore mined in a 2.4 year period -- $1.22 \times 10^6 m^3$ and $2.8 \times 10^5 m^3$ at the average large mine and average mine, respectively (the bulking factor is not considered in computing the pit volume).

$$V \text{ (ore + sub-ore zone)} = 1/3 \; \pi h_1 \; (r_1^2 + r_1 r_2 + r_2^2) \qquad (3.4)$$

The computed radii, r_1 and r_2, were 174 m and 186 m at the average large mine and 80 m and 92 m at the average mine. The surface areas (S_A) of exposed ore and sub-ore in the pit are then one-half that given by the equation,

$$S_A = 1/2 \pi \; (d_1 + d_2)(\text{slant height}) + \pi r_1^2, \qquad (3.5)$$

where d_1 and d_2 are the diameters related to r_1 and r_2. Exposed surface areas of ore and sub-ore were computed to be equal and 57,170 m^2 at the average large mine and 14,650 m^2 at the average mine.

The shape of the overburden zone was assumed to be the same as the ore and sub-ore zone (Fig. 3.14). The thickness, h_2, and radius, r_3, of this zone can be computed using the following equation with the relationship, $r_3 = r_2 + h_2$, and knowing the volume--$4.8 \times 10^7 m^3$ and $7.2 \times 10^6 m^3$--at the average large mine and average mine, respectively.

$$V \text{ (overburden)} = 1/3 \; \pi h_2 \; (r_2^2 + r_2 r_3 + r_3^2) \qquad (3.6)$$

Since r_2 was computed above to be 186 m at the average large mine and 92 m at the average mine, Equation 3.6 becomes

$$4.8 \times 10^7 = 1.087 \times 10^5 h_2 + 584 h_2^2 + 1.047 h_2^3 \qquad (3.7)$$

for the average large mine, and
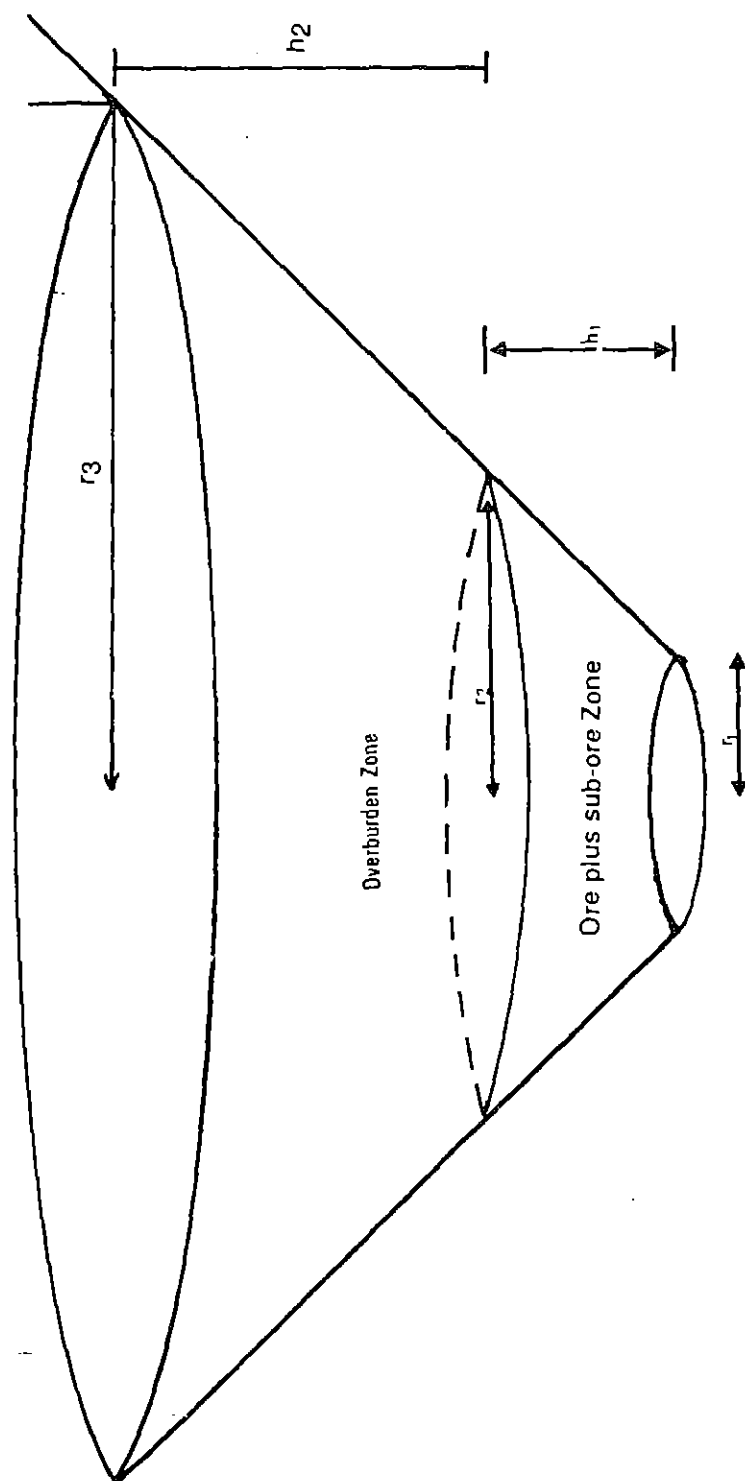
Figure 3.14 Configuration of open pit model mines.

$$7.2 \times 10^6 = 2.659 \times 10^4 h_2 + 289 \ h_2{}^2 + 1.047 h_2{}^3 \qquad (3.8)$$

for the average mine.

Solving these equations yields the following parameters:

	h_2	r_3	r_2
average large mine	188 m	374 m	186 m
average mine	105 m	197 m	92 m

The surface area (S_A) of the exposed overburden is then given by the following equation.

$$S_A = 1/2 \ \pi \ (d_2 + d_3) \ (\text{slant height}), \qquad (3.9)$$

where d_1 and d_2 are the diameters related to r_2 and r_3. Areas computed were $4.68 \times 10^5 m^2$ and $1.34 \times 10^5 m^2$ for the average large mine and average mine, respectively.

Multiplying the exposed ore, sub-ore, and overburden areas by their U_3O_8 contents (0.10%, 0.015% and 0.002%, respectively) and by a Rn-222 exhalation rate of 0.092 Ci/m^2 per year per percent U_3O_8* and summing gives the annual Rn-222 releases shown in Table 3.35.

The emanation of Rn-222 from overburden, sub-ore, and ore storage piles is, based on an exhalation rate of 0.092 Ci/m^2 per yr per percent U_3O_8 (Ni79), and ore grades of 0.002 percent, 0.015 percent, and 0.10 percent, respectively. The surface areas used were those computed previously for the case 2 model mines and listed in Tables 3.11, 3.17 and 3.20. The areas for the average large mine and average mine are $1.1 \times 10^6 m^2$ and $2.2 \times 10^5 m^2$ for overburden piles, $1.2 \times 10^5 m^2$ and $3.6 \times 10^4 m^2$ for sub-ore piles, and $6.2 \times 10^3 m^2$ and $3.6 \times 10^3 m^2$ for the ore piles, respectively. Applying these parameters, the annual Rn-222 emissions from the overburden, sub-ore, and ore piles at the average mine and average large mine were computed. Table 3.35 presents the results.

The total annual Rn-222 released during surface mining operations is the sum of the releases from the sources considered: 331 Ci from the average mine and 1261 Ci from the average large mine. Considering ore production and

---

*The average value of measured exhalation rates at surface uranium mines (Ni79).

Table 3.35   Radon-222 releases during surface mining, Ci/yr

Source	Average Mine	Average Large Mine
Ore loading and unloading	9	39
Reloading ore from stockpile	9	39
Sub-ore loading and unloading	1	6
Overburden loading and unloading	9	61
Exposed surface of overburden, ore, and sub-ore in the pit	180	691
Ore stockpile exhalation	33	57
Sub-ore pile exhalation	50	166
Overburden pile exhalation	40	202
Total	331	1261

grade differences, these values agree reasonably well with those computed by other procedures (Tr79).

3.4   Underground Mining

3.4.1   Solid Wastes

During underground mining, like surface mining, materials are removed, separated according to ore content, and stored on the surface for various periods of time (Section 1.3.3). These separate piles consist of waste rock produced from shaft sinking operations and from cutting inclines, declines, and haulage drifts through barren rock, sub-ore, and ore. The waste rock is similar to overburden removed at surface mines, except much smaller quantities are involved and none are returned to the mine. However, as mining progresses, waste rock is sometimes used to backfill mined out areas of the mine and retained beneath the surface. The ore and sub-ore will also be similar in nature to those described previously for surface mines, as is their potential to be sources of contamination to the environment (Fig. 3.15).
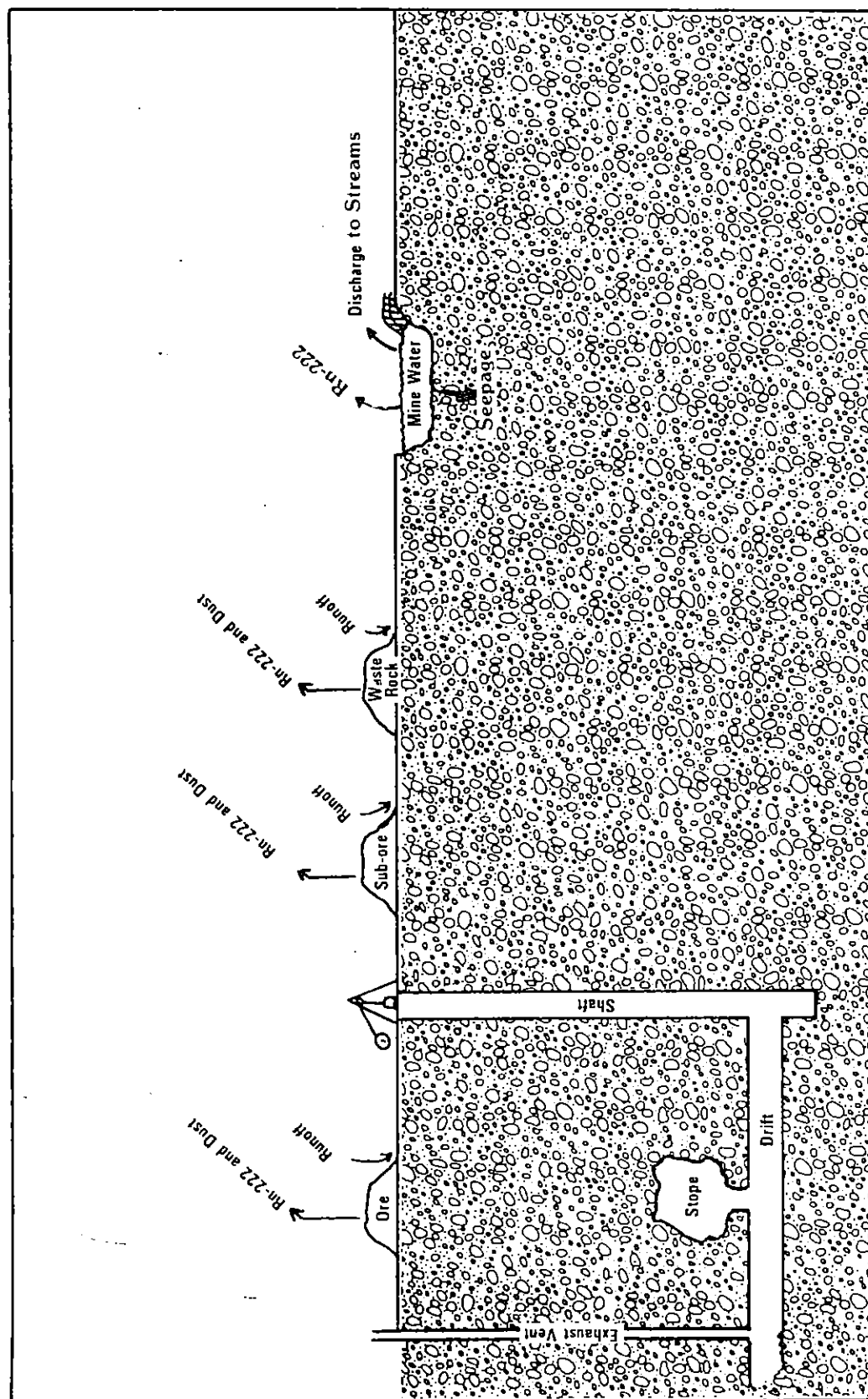
Figure 3.15 Potential sources of environmental contamination from active underground uranium mines.

3.4.1.1 Waste Rock Piles

Much smaller quantities of waste rock accumulate at underground mines than overburden at surface mines. The weight ratio of waste rock to ore depends mainly upon the size, depth, and age of the mine. During the initial mining stages, all material removed is waste rock. As entry into the ore body occurs and ore mining begins, the quantity of waste rock removed per metric ton of ore decreases sizably. Once in the ore body, as little waste rock as possible is mined. The ratio of ore to waste rock removed from underground mines varies considerably. At seven presently active underground mines, the ore to waste rock ratio varies from 1.5:1 to 16:1, with an average ratio of 9.1:1 (Jackson, P.O., Battelle Pacific Northwest Laboratory, Richland, WA, 12/79, personal communication). As future mines become larger and deeper, the overall ore to waste rock ratio will probably decrease.

Since the annual average ore capacity of underground mines was 1.8×10^4 MT in 1978 (Section 1.3.1), the average of the 305 underground mines would have produced 2.0×10^3 MT of waste rock during that year, assuming the average 9.1:1 ore to waste rock ratio. This will be considered the production rate of the "average underground mine." Like surface mines, relatively few of the 305 active underground mines account for a significant portion of the total ore produced by the underground method. Also, future underground mines are expected to have larger capacities than many of the current mines (Th79). Therefore, a second underground mine will be considered, which is defined as the "average large mine." Its annual ore production rate is assumed to be 2×10^5 MT, the average ore capacity of five large underground operations (Ja79b, TVA79, TVA76, TVA78a, TVA78b). The quantity of waste rock removed annually will be 2.2×10^4 MT, assuming the ore to waste rock ratio to be the same as for the average mine. Assuming the density of waste rock to be about 2.0 MT/m^3 and a bulking factor of 1.25 (Burris, E., Navajo Engineering Construction Authority, Shiprock, N.M., 2/80, personal communication), the average mine and average large mine will produce waste rock at an annual rate of 1.3×10^3 m^3 and 1.4×10^4 m^3, respectively. Since waste rock is not presently used to backfill mined-out areas, this rate of accumulation will continue for the life of the mine, which is assumed to be the same as that for an open pit mine, 17 years.

Table 3.36 lists estimated average surface areas of the waste rock piles during the lifetimes of the two mines defined above. The following parameters were used:

Parameter	Average Mine	Average Large Mine
Waste rock production rate, MT/yr	2.0×10^3	2.2×10^4
Rock density, MT/m^3	2.0	2.0
Bulking factor	1.25	1.25
Waste rock volume, m^3/yr	1.3×10^3	1.4×10^4
Active mine life, yr	17	17
Pile height, m	6	12

These estimated areas assume no backfilling and that the piles are on level terrain. Because waste rock is sometimes used to backfill and is often dumped into a gorge or ravine, these surface areas represent maximum conditions.

The mineralogy, physical characteristics, and composition of waste rock from underground mines are assumed to be identical to the overburden removed from open pit mines (Section 3.3.1.1). Also, reclamation procedures for waste rock piles at underground mines should be similar to those described in Section 3.3.1.4 for overburden dumps.

3.4.1.2 Ore Stockpiles

Because ore is often stockpiled at the mine and/or at the mill, it becomes a potential source of contamination to the mine environment during the storage period. These piles will be smaller than the waste rock piles, but the concentration of most contaminants in the ore-bearing rock will be much greater.

Ore stockpile residence times can vary considerably with time and ore management. Residence times commonly range from a few days to a few months. The same residence time will be assumed for underground mines as was selected above for surface mines, 41 days. Assuming a 330 operating-day-year and a 1.25 bulking factor, the ore stockpiles of the average mine and average large mine will contain 1,400 m^3 and 15,500 m^3 of ore, respectively. The surface areas of the ore stockpiles were computed using these volumes and assuming 3.1 m high rectangular piles (NRC78a). Table 3.37 lists the estimated surface areas.

Table 3.36 Estimated average surface areas of waste
rock piles at underground mines

Mine Size	Average Accumulation,[a] m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine[b]	1.1×10^4	2,700	2,460
Average large mine[c]	1.2×10^5	14,100	12,800

[a]Assumes average volume of waste rock accumulated during 17-yr. mine life with no backfilling (1/2 total volume accumulation).

[b]Annual waste rock production = 2.0×10^3 MT.

[c]Annual waste rock production = 2.2×10^4 MT.

Note.--Waste rock piles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8 a).

Table 3.37 Estimated surface areas of ore stockpiles
at underground mines

Mine Size	Steady State Accumulation,[a] m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine[b]	1,400	680	620
Average large mine[c]	15,500	5,800	5,480

[a]Assume 41-day residence time.

[b]Annual ore production = 1.8×10^4 MT.

[c]Annual ore production = 2×10^5 MT.

Note.--Ore stockpiles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8 a). Pile height is assumed to be 3.1 m (NRC78a).

The mineralogy, physical characteristics, and composition of ore from underground mines are assumed to be identical to the ore removed from surface mines (Section 3.3.1.2). The U_3O_8 grade of ore may average somewhat higher from underground mines than from surface mines. However, a grade of 0.1 percent U_3O_8 probably approximates reasonably well the ore reserves minable by the underground method (DOE79). Uranium and its decay products in airborne dust from these ore piles will be concentrated by a factor of 2.5 (Section 3.3.1.2).

3.4.1.3   Sub-Ore Piles

The quantity of sub-ore mined at an underground mine, as at a surface mine, is considered to be about equal to the quantity of ore mined, 1.8×10^4 MT at the average mine and 2×10^5 MT at the average large mine. Assuming sub-ore to have a density of 2.0 MT/m^3 and after removal a bulking factor of 1.25, the average volume of sub-ore to be on the surface during the 17-yr operational life of the average mine and average large mine will be 9.6×10^4 m^3 and 1.1×10^6 m^3, respectively (i.e., one-half the total of 17-yr accumulation).

Although sub-ore is often placed on top of piles of previously mined waste rock (Perkins, B.L., New Mexico Energy and Minerals Department, Santa Fe, NM, 12/79, personal communication), we assumed separate rectangular piles in computing the surface areas of the piles at the model mines. Table 3.38 lists the estimated surface and pad areas of the sub-ore piles. These computations were based on pile heights of 6 m at the average mine and 12 m at the average large mine.

At underground mines, the cutoff grade ranges from 0.02 to 0.05 percent U_3O_8, yielding an average sub-ore grade of 0.035 percent U_3O_8 (99 pCi/g) (Perkins, B.L., New Mexico Energy and Minerals Department, Santa Fe, N.M., 12/79, personal communication). The mineralogy, physical characteristics, and other constituents of sub-ore from underground mines are assumed identical to the sub-ore removed from surface mines (Section 3.3.1.3).

Table 3.38   Estimated average surface areas of sub-ore
piles at underground mines

Mine Size	Average Accumulation,[a] m^3	Surface Area of Pile, m^2	Surface Area of Pad, m^2
Average mine[b]	9.6×10^4	18,800	17,700
Average large mine[c]	1.1×10^6	104,900	99,400

[a]One-half that which will accumulate during the 17-yr mine life.

[b]Annual sub-ore production = 1.8×10^4 MT.

[c]Annual sub-ore production = 2.0×10^5 MT.

Note.--Sub-ore piles are rectangular with length twice the width and sides sloping at 45° (Fig. 3.8a).

3.4.2   Mine Water Discharge

3.4.2.1   Data Sources

Information concerning the amount and quality of water discharged from underground uranium mines in New Mexico is from field surveys conducted in 1975 (EPA75, P. Frenzel, USGS, written communication, 1979) and Wogman (Wo79), from site-specific environmental impact statements and reports, from NPDES permits, and from a State study (Pe79).

Many mining companies maintain that permits are not required because the formerly ephemeral streams into which discharge occurs are, in effect, a result of the discharges and do not meet the definition of navigable bodies of water. Nevertheless, the companies have applied for permits, together with a request to the courts for a ruling concerning their necessity.

The New Mexico district office of the U.S. Geological Survey (L. Beal, USGS, written communication, 1979) provided discharge rate and volume for the regional drainage systems, namely the Rio San Jose, Rio Puerco (east), and the Rio Grande. We followed procedures developed by the USGS (Bo70) to calculate runoff from ungaged basins.

3.4.2.2 Quality and Quantity of Discharge

To estimate average or typical conditions for mine water discharge, 11 projects in Colorado, New Mexico, and Utah were selected. Table 3.39 shows the summarized flow and water quality data. The center of current domestic underground mining is in the Colorado Plateau and the San Juan Basin. In this area, there is an increasing trend toward underground mining. In Wyoming, both underground and surface mining activity are significant. In Texas, surface mining and, to a lesser extent, in situ leaching are the principal methods used. Climatic and geologic characteristics and land and water use patterns in the Colorado-Utah-New Mexico uranium area are broadly similar; and the Grants Mineral Belt in general and the Ambrosia Lake District in particular are representative of this area. There are many complicating variables such as the geologic and geochemical characteristics of the ore body and host rock. Water-yield and quality associated with mines also vary within the region, as do the size and relative location of the populace. The Grants Mineral Belt scenario is conservative. The mines discharge relatively large amounts of water to streams that are used for irrigation and stock watering and that flow by or through local centers of population.

Table 3.39 shows discharge from selected underground uranium mines in the Colorado Plateau areas of Colorado, New Mexico, and Utah. On the average, discharge is 2.78 m^3/min, with a standard deviation of 4.34 m^3/min. The selected underground mines discharge an amount of water similar to that from the Wyoming surface mines. In the Grants Mineral Belt area, average flow from 28 underground mines is 2.4 m^3/min (J. Dudley, New Mexico Environmental Improvement Division, written communication). Of the 27 active underground mines being dewatered, 17 discharge to the environment at an average rate of 3.2 m^3/min. The remainder are in a closed circuit. That is, their discharge is used as mill feed water. The range for 17 mines is 0.2 to 19 m^3/min. Average discharges from New Mexico underground mines are significantly greater than those from mines in Colorado and Utah, which average 0.68 m^3/min. Most of the ore production in New Mexico has been from mines 200 to 300 meters deep. In recent years, mines have become progressively deeper and involve more dewatering. For example, the Gulf Mount Taylor mine, which is not yet producing ore, discharges 15 m^3/min and will produce ore from a depth

of 1,200 meters. Most of the water is now diverted to a nearby ranch for irrigation and stock watering. When the mill goes on line, most of the mine water will be used there.

Of the 16 active mines in the Ambrosia Lake district, 13 discharge to offsite areas at an average rate of approximately 1.6 m^3/min. For modeling and to be conservative, we assumed that 14 active mines are present in the model mine area and that the average discharge rate per mine is 2.0 m^3/min. This is somewhat less than the average condition for the Grants Mineral Belt (3.2 m^3/min) as a whole in terms of discharge rate, but the high density of mines assumed present in the model area partly compensates for the difference.

For the New Mexico project shown in Table 3.39, numbers 4, 5, 6, and 7 have discharge that comes directly from the mine portal to settling ponds before discharge. Neither ion exchange for uranium recovery nor barium chloride treatment for radium removal is used. Facilities 8 through 11 use ion exchange columns for uranium removal before discharge. Settling may or may not be used, depending on the suspended solids content of the particular discharge. Project number 10 removes radium prior to discharge. Radium concentrations in the combined effluent from two active mines in the Church-rock area (projects 8 and 4), both of which use settling ponds as the only treatment, have ranged from 1.9 to 8.9 pCi/ℓ since 1975. In the first survey (EPA75), effluent from these same mines contained 30.8 and 7.9 pCi/ℓ. The combined discharge from both mines was sampled by the U.S. Geological Survey in 1975, 1977, and 1978 (P. Frenzel, written commu/nciation) and by the EPA (EPA75) in 1975. Concentrations were 30, 14, 2.6, and 2.6 pCi/ℓ, respectively.

It is apparent that there are marked temporal trends in mine water quality and quantity. Major factors responsible include changes in the dewatering rate accompanying shaft sinking versus actual ore production. Simultaneously, there are changes in the mineral quality and leaching rate of strata as the ore body is approached and then penetrated. Mining practices, oxidation of the ore body and possibly bacterial action may also assist in the solubilization of toxic stable and radioactive trace elements. Sample handling and analytical procedures can also markedly affect results. For example, if suspended solids are high and a sample is acidified prior to filtering, soluble radium, uranium, and other trace constituents typically

Table 3.39   Summary of average discharge and water quality data for underground uranium mines in the Colorado Plateau Region (Colorado, New Mexico, Utah) and a comparison with NPDES limits

Project	Discharge m³/min	Dissolved Radioactivity			Major and trace constituents, mg/ℓ							
		Total U, mg/ℓ	Ra-226, pCi/ℓ	Pb-210, pCi/ℓ	TSS	SO₄	Zn	Ba	Cd	As	Mo	Se
Utah 1(a)	0.67	1.35	1.25		7.5							
Colorado												
2	1.31	2.20	0.53		14.3	872	0.02	0.19	< 0.01	< 0.01	0.4	0.03
3	0.06	0.25	10.00		144.9		0.065		0.003	0.055	0.054	0.008
New Mexico												
4	14.67	1.0	89(b)	15	25.4	60.6		2.13		<0.005	< 0.01	0.004
5	3.79	0.67	23(c)	33	2.6	213.7				0.011	0.24	0.002
6	1.89	0.02	14(c)	15	51.5	744				0.005		0.094
7	0.95	0.18	0.1	0		1045				<0.005	0.05	0.407
8(a)	0.18	4.2	1.9	9.7		67.2		0.88		<0.005	< 0.01	0.027
9(a)	0.82	1.9	4.7	16	1	675		0.17		0.011	0.45	0.036
10(a)	6.06	1.1	2.3	14	1.08	705				< 0.005	0.62	
11(a)	0.216	2.6	4.3	14	2.2	837		0.66		0.012	0.79	
Average	2.78	1.41	13.7	14.6	27.8	580		0.81		0.012	0.29	0.076
Standard Deviation	4.34	1.25	25.9	9.1	46.9	368		0.80		0.015	0.29	0.137

Table 3.39 (continued)

Summary of NPDES permit limits for daily average/daily maximum, mg/ℓ except Ra-226, pCi/ℓ

State	Dissolved Radium-226	Dissolved Uranium	Total Suspended Solids	Total Dissolved Solids	Zinc	Barium	Cadmium	Arsenic	Vanadium
New Mexico	3/10 10/30 Total Radium	2/4	50/150(day)[d] 20/30(month)		0.5/1.0				
Utah	3/10 and -/3	2/4 and -/2	20/30[e]	NA/650 3500	0.5/1.0				
Colorado	3/10 and -/3	3/5 and 2/4	20/30	48990/[f] 122476 kg/day	0.5/1.0	1/2 and -/1	0.05/0.1	1/2 and 0.5/1	5/10

[a] Average discharge rate per mine is shown. Two or more mines constitute the project.

[b] $BaCl_2$ treatment for radium removal faulty; repaired in late 1979.

[c] Values shown are for untreated water. $BaCl_2$ treatment now used.

[d] Applies to discharge associated with shaft construction.

[e] Maximum of 10 mg/ℓ for 30-day period and 20 mg/ℓ for 7-day period effective July 1, 1980.

[f] Receiving water standard.

Source: Chemical analyses from in-house studies (EPA75) and State of New Mexico (J. Dudley, Environmental Improvement Division, written communication). NPDES permit data from Regions VI, VIII (H. May, R. Walline, written communication). Other references include site-specific reports (EIS,ER) and company monitoring data.

will increase, as compared to samples that are filtered prior to acidification (Ka77). Therefore, development of "average" or "typical" trace element concentration data is questionable and may be erroneous without detailed knowledge of the many variables affecting the final results.

Despite the foregoing difficulties, available chemical data assembled in Table 3.39 provide much of the source term input data used in subsequent calculations. The reader should bear in mind that uranium concentrations are likely to be less than 3 mg/ℓ simply because it is economically practical to use ion exchange recovery for concentrations greater than this level. Daily average radium-226 concentrations on the order of 3 pCi/ℓ are specified in valid NPDES permits, and reliable data from USGS, EPA, and state sources reveal stream concentrations near the point of discharge to be on the order of 3 to 14 pCi/ℓ in recent years. Therefore, the "average" radium-226 concentration of 13.7 pCi/ℓ used in the subsequent modeling calculations is at least slightly conservative. Actual concentrations of stable elements (Zn, Ba, Cd, etc.) appear to be well below the NPDES limits, which were also developed from analysis of uranium mine effluent. Thus, it is presumed that the average values in Table 3.39 for these elements are reasonably correct. The variables of mine size, age, host rock, and water treatment (ion exchange, barium chloride, settling ponds) are reflected in the data. Water quality for mines examined in Utah and Colorado generally agrees with the New Mexico cases, with the exception of Project Number 3 mine, which is being dewatered and may, therefore, temporarily have excessive suspended solids. We recommend that the NPDES data for uranium mine discharges be evaluated and that additional compliance monitoring be conducted to confirm the quality of mine discharge. Such studies should focus on situations where mine water is being used for irrigation and stock watering.

Table 3.40 shows discharge and water quality characteristics for underground mines under construction and not yet producing ore. The first example involves water pumped from a deep mine shaft under construction. Considerable water is encountered above the ore body; water quality is good and representative of natural conditions; and suspended solids are high as a result of construction. The second case is similar except that flow is reduced, but radium and suspended solids concentrations are greatly elevated due to construction and possible ore body oxidation. The third case involves

Table 3.40 Water quality associated with underground mines in various stages of construction and operation

| Project | Discharge m³/min | Dissolved | | | TSS | Concentration, mg/ℓ | | | |
		Total U mg/ℓ	Ra-226 pCi/ℓ	Pb-210 pCi/ℓ		SO$_4$	As	Mo	Se
New Mexico									
1. Underground mine shaft construction; dewatering	5.76	0.03	0.07	10	23.8	134	<0.005	0.01	0.003
2. Underground mine shaft construction; dewatering	1.73	<0.01	29	0	554	527	0.012	0.007	0.005
3. Underground mine; dewatering wells	1.43	0.08	0.2	0	1	144	<0.005	0.01	0.003
4. Underground mine recirculating leach solution from stopes (after ion exchange)	0	0.32	29	17	1.1	1060	<0.005	3.2	0.268

Source: J. Dudley, State of New Mexico, written communication, 1979.

dewatering wells used to dewater the ore body before mining. There is no oxidation and suspended solids are very low as is radium-226. Dissolved radium-226 in the ore body is on the order of 10 pCi/ℓ or less in the natural state, but concentrations rise to 100 pCi/ℓ or more after mining takes place, possibly due to oxidation and bacterial action in the workings (EPA75). Project Number 4, in the Ambrosia Lake district, is an inactive underground mine now used as a type of in situ leach facility. Mine water is recirculated through the workings. Leached uranium is selectively recovered using ion exchange. The process is a closed one, hence no effluent is involved. Water quality after uranium removal reflects the buildup in radium, lead-210, sulfate, molybdenum, and selenium. ···

3.4.3. Hydraulic and Water Quality Effects of Underground Mine Discharge

3.4.3.1 Runoff and Flooding in the Model Underground Mine Area

3.4.3.1.1 Study Approach

We chose to study an area of rather concentrated underground mining, similar to the Ambrosia Lake district of New Mexico. All of the mines in the district dewater to different degrees because the principal ore body is in the Westwater Canyon Member of the Morrison Formation, which is also a major aquifer. In the analysis, flows from some 14 active mines discharge to formerly dry washes and dissipate downstream by evaporation and, more importantly, infiltration. Suspended and dissolved constituents persist at the land surface and become available for resuspension and transport in surface floods with recurrence intervals of 2 to 25 years. Contaminated runoff from the sub-basin is then diluted in average annual flows of progressively larger streams and rivers of the region.

Similar to the analysis presented for surface mines in Wyoming, there is a three-basin hierarchy: sub-basin, basin, and regional basin (Fig. 3.16). These correspond to Arroyo del Puerto-San Mateo Creek, Rio San Jose and Rio Puerco, and the Rio Grande. Of these, the Rio Puerco is distinctly ephemeral. The Rio Puerco drains into the Rio Grande, which is perennial, due in large part to the heavily regulated flows and storage reservoirs. Because

MCKINLEY COUNTY

3-121

SANDOVAL COUNTY

Sub-basin area containing model mines

Grants

Rio

3435

San Jose

Rio Puerco

BERNALILLO COUNTY

N

0   10   20   30   40   50
         (km)

VALENCIA COUNTY

Rio

3530

3320

Grande

.... Sub-basin boundary
—·— Boundary of a portion of the Rio San Jose basin
         upstream from the sub-basin
—·—·— Rio Puerco basin boundary
▲ USGS gauging station
Note: Boundary of the Rio Grande regional basin above Bernardo is not
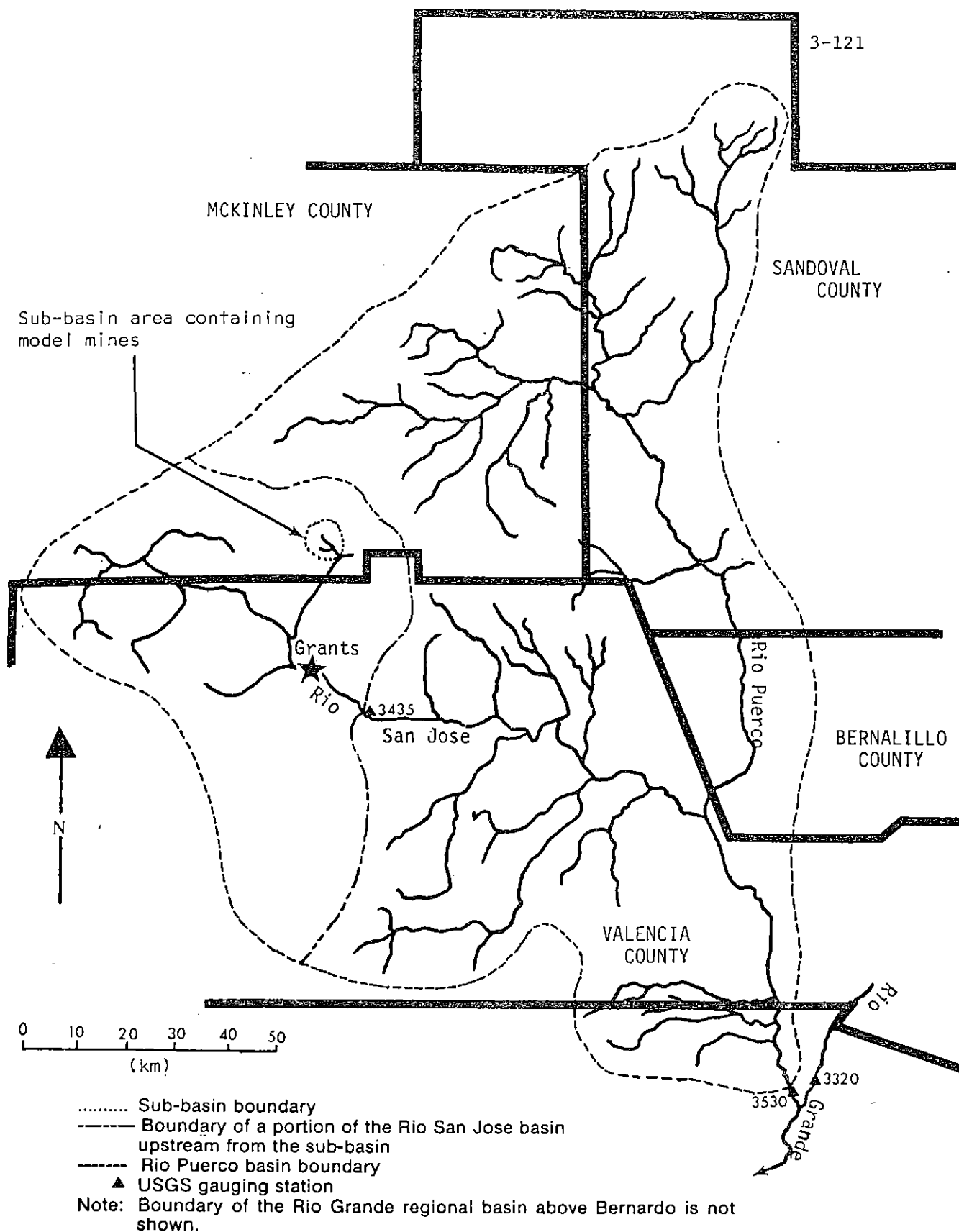   shown.

Figure 3.16   Sketch of sub-basin, basin, and regional basin showing orienta-
                   tion of principal drainage courses, areas of drainage, and loca-
                   tion of mines in the New Mexico model area

the Rio Grande is the major regional river and the basis of extensive irrigation projects, it is included in the analysis. The mining area is well away from the Rio Grande Valley, and it is unlikely that noticeable changes in flow or water quality because of mining would occur.

Flow volumes for the sub-basin and open file USGS data (L. Beal, written communication, 1979) for flows in the basin and regional basin are used to transport and dilute contaminants originating in the mine effluent. It is initially assumed that all contaminants are available for transport by surface flow so as to deliberately create a worst-case situation. Section 3.4.3.2 reviews infiltration of water and solute for possible effects on groundwater.

We do not address the effects of seepage from settling ponds because such ponds are relatively small, tend to be self-sealing, and are well away from inhabited areas. Supposedly, settled solids from these ponds are removed and incorporated with uranium mill tailings. Limited field studies to determine whether such ponds cause groundwater contamination is warranted. In some instances, the ponds have synthetic liners, and leakage is expected to be minimal. The influence of mine dewatering (by wells, shafts, and pumping of mine workings) on groundwater quality or availability is not addressed primarily because of the lack of data. We strongly recommend further study of the hydraulic and groundwater quality effects of dewatering. This aspect of mining is coming under increased scrutiny by regulatory agencies at the State and Federal level because of the influence on water quality and availability.

In summary, our approach defines the quality and volume of mine water discharge; outlines hydrographic basins; and calculates flood flows for various return periods ranging from 2 to 25 years in the sub-basin. These flows are then diluted into the average annual flow in the basin and regional basin. The principal objective is to develop a rough estimate of contaminant loads resulting from mine discharge.

3.4.3.1.2 Description of Area

The Grants Mineral Belt of northwestern New Mexico is in the Navajo and Datil sections of the Colorado Plateau physiographic province (Fe31). Characteristic landforms in the study area include rugged mountains, broad, flat

valleys, mesas, cuestas, rock terraces, steep escarpments, canyons, lava flows, volcanic cones, buttes, and arroyos (Ki67; Co68). Elevations in the area range from 1,980 m at Grants to an average of 2,160 m near Ambrosia Lake. Just north of Grants is Mount Taylor, the highest point in the region. It rises from Mesa Chivato to an elevation of 3,471 m (Co68).

The study area has a mild, semiarid, continental climate. Precipitation averages 25.4 cm/year, and there is abundant sunshine, low relative humidity, and a comparatively large annual and diurnal temperature range. Average annual precipitation at Gallup, Bluewater, and Laguna is 27.12, 24.55, and 22.31 cm, respectively. In the higher elevations, the average is 51 cm or more because of thunderstorms in July, August, and September and snow accumulations in the winter months (Co68, Go61, Jo63). Only thunderstorms are significant in the lowlands. Heavy summer thunderstorms (40 to 70 in number) of high intensity and local extent can result in 5 cm of rain with local, damaging flash floods.

The watersheds of the Rio San Jose and Rio Puerco encompass 19,037 km^2. Most of the larger communities in the basin are located in the floodplain of the Rio Grande and principal tributaries. Extensive irrigation with surface water occurs in the watersheds of the Rio San Jose, Rio Puerco, and Rio Grande. In the sub-basin, there was no perennial flow before mining and, thus no irrigation, but increasing use is being made of the mine discharge, which is regarded as an asset in a water-short area. Subsequent sections summarize the surface water quantity at some of the principal gauging stations in the Middle Rio Grande Basin and the irrigated areas below these stations. Groundwater is used for essentially all public water supplies as the temperature, quality, and year-round availability are assured. Numerous wells scattered across the landscape, particularly in the stream valleys, are used for stock water and, to a lesser extent, for potable use on the scattered ranches and Indian settlements.

Under completely natural conditions, streams in the study area were distinctly ephemeral, and many of the smaller ones did not experience flow for periods of several years. The Rio Grande experiences peak flows in the April-June period when snowmelt and precipitation cause gradual rises to moderate discharge levels involving large volumes of flow and long durations. Peak discharge rates (volume per time) occur in the summer flash floods.

Construction of dams and conveyance channels to eliminate flooding problems has been extensive. In the tributaries such as the Rio San Jose and upper reaches of the Rio Puerco, there is considerable streamflow regulation to minimize flood damage and maximize use of available water for irrigation.

Conditions in the Ambrosia Lake district with respect to the type of mining operations and discharge of effluent to ephemeral streams are duplicated elsewhere in the Grants Mineral Belt. In the Churchrock district, two mines discharge to the Rio Puerco at rates of 4.7 to 15 m^3/min. Most of the 4.7 m^3/min discharge from one mine is now used in a nearby mill. At Mariano Lake, located between Ambrosia Lake and Churchrock, and at the Marquez and Rio Puerco mines east of Ambrosia Lake, mines are expected to discharge 0.8 to 4.5 m^3/min to various ephemeral streams. Another large mine will soon discharge up to 5.3 m^3/min northward into the San Juan River Basin. In the mid 1980's, construction is expected to begin on five large underground mining projects that will have a combined discharge on the order of 71 m^3/min. Most discharge will be into the San Juan Basin, reflecting the trend of mines becoming deeper and requiring more dewatering as the mining center moves from the south flank of the San Juan Basin into more interior portions.

3.4.3.1.3   Estimate of Sub-basin Flood Flow

Since we use a dilution-model, emphasis is on flow volume rather than peak discharge rate in the sub-basin, basin, and regional basin hydrographic units. Gaging records from the U.S. Geological Survey WATSTORE system (L. Beal, written communication, 1979) provide average discharge rates for runoff events with various return periods and durations. The latter specify the time, in days, and the associated flow rate that will be equaled or exceeded. Flows for arbitrary periods of time ranging from 1 to 183 days are specified. Probability can be stated in terms of N-year recurrence interval. By combining discharge rate (volume per time) and time (partial duration), flow volume can be calculated.

In the ungaged sub-basin, runoff volumes associated with events having return periods of 2, 5, 10, and 25 years were calculated from regression equations developed by the USGS (Bo70). The equations were generated from multiple regression of discharge records from gaged basins against various basin characteristics. These are area (A), precipitation (P_a), longitude

at the center of the sub-basin (Lo), soils infiltration index (Si), and mean basin elevation (E_m). Through use of appropriate constants and coefficients (Bo70), flow volumes can be calculated for 1-day and 7-day events with return periods of 2, 5, 10, and 25 years. For the sub-basin, the basic equation has the following form:

$$FV = a \; A^{b_1} \; P_a^{b_9} \; Lo^{b_{14}} \; S_i^{b_{15}} \; E_m^{b_4} \qquad (3.10)$$

where
$A = 95 \; mi^2$
$P_a = 2.9$ inches
$Lo = 7.85$ (longitude in decimal degrees minus 100)
$S_i = 8.5$
$E_m = 7.0$ thousand feet

Table 3.41 contains the regression coefficients and total flow volume data. Short-term, 1-day and 7-day, events were of main interest because these would be expected to provide greater flushing of contaminants stored at or near the water-substrate interface in the streams receiving mine discharge.

The extent to which mine discharge transforms existing ephemeral streams into perennial ones is evaluated with a crude seepage and evaporation model (see Appendix H). The basic equations and approach are patterned after a similar analysis in the Generic Environmental Impact Statement on Uranium Milling (NRC79b).

Figure 3.16 shows the relationship of the sub-basin, basin, and regional basin boundaries and the principal drainage courses and gaging stations. The confluence of the Rio Puerco and Rio San Jose is shown approximately 55 km closer to the Rio Grande than is actually the case in order to simplify flow routing and to reduce the number of dilution calculations. Table 3.42 summarizes the key characteristics of these basins in terms of catchment area, discharge, and irrigated farmlands downstream from points where mine discharge might be tributary to the streams. Mine discharge occurs in the sub-basin which in turn discharges to the Rio San Jose and then to the Rio Puerco. No mine discharge and no significant runoff are associated with that portion of the basin tributary to San Mateo Creek between the Rio San Jose and the sub-basin. For modeling, flooding within and runoff from the sub-

Table 3.41　Total flow volume for sub-basin floods of 1- and 7-day durations and return periods of 2, 5, 10, and 25 years

Flood Volume	a	Regression Coefficients					Volume (m^3)
		b_1	b_9	b_{14}	b_{15}	b_4	
FV 1,2[a]	1.08×10^{-4}	0.931	1.83	-1.43	4.09	–	2.16×10^4
FV 1,5	1.27×10^{-3}	0.941	1.40	-1.89	4.07	–	6.23×10^4
FV 1,10	5.07×10^{-3}	0.953	1.17	-2.18	4.02	–	1.02×10^5
FV 1,25	2.39×10^{-2}	0.972	0.929	-2.51	3.95	–	1.76×10^5
FV 7,2	8.60×10^{-7}	0.965	2.36	-1.61	4.22	1.50	5.95×10^3
FV 7,5	2.99×10^{-4}	0.904	2.55	-2.09	3.53	–	8.79×10^3
FV 7,10	8.97×10^{-4}	0.910	2.37	-2.39	3.61	–	1.43×10^4
FV 7,25	3.06×10^{-3}	0.922	2.17	-2.76	3.68	–	2.26×10^4

[a] FV 1,2 indicates a flood of 1-day duration and a return period of 2 years.

Table 3.42   Summary of area, discharge, and irrigated acreage for the sub-basin, basin, and regional basin hydrographic units in New Mexico

	USGS Station Number	Area (km^2)	Period of Record Yrs.	Number of km^2 Under Irrigation Below Station	Average m^3/min Discharge (for Period of Record)	Average Annual Discharge (m^3) for Period of Record
Sub-basin	---	246	---	---	---	---
Basins						
Rio San Jose near Grants	3435	5957 (2927 non-contributing)	42	2.43+	11.09	5.83×10^6
Rio Puerco near Bernardo	3530	19037 (2927+ non-contributing)	38	---	81.05	4.26×10^7
Regional Basin						
Rio Grande at Bernardo	3320	49810 (7610 non-contributing)	41	N/C[a]	1649.35	86.69×10^7

[a] N/C = Not Calculated.

basin is, in effect, routed without change in flow and quality and allowed to enter the Rio San Jose. Flow from the San Jose is further diluted in the Rio Puerco, then diluted again in the Rio Grande. In actuality, flow from the Ambrosia Lake district rarely, if ever, enters the Rio San Jose because flood volumes are small and infiltration losses are large. This departure from true conditions is justified within the context of the modeling approach used. Basically, the model draws from a specific area but does not attempt to closely duplicate its conditions. If a specific area were exactly represented, the model would still be incorrect to varying degrees for other areas, and the generic value of the assessment would depreciate.

Of special interest is the effect of contaminated flows on irrigation projects present on the Rio San Jose and Rio Grande. An extensive system of dams and conveyance channels regulates flow in the Rio Grande, and partial duration flow data are unavailable. Instead, the average annual flow volume is used to provide the final dilution estimate. For the sub-basin in which the mines are located, flood volumes are calculated using the USGS regression equations (Bo70). The maximum return period for which flows are calculated is 50 years. The remainder of this section first considers the flow or hydraulic aspects of the surface water pathway. Finally, several factors concerning the quality of runoff water are mentioned to balance conservatism and realism in the pathway analysis and, subsequently, in the health effects modeling to follow. The emphasis here is on surface water impacts, and we assume maximum transport for this pathway. The influence of infiltrating mine water is discussed in Section 3.4.3.2.

All of the streams, except the Rio Grande and certain reaches of the Rio San Jose are distinctly ephemeral under natural conditions. In the sub-basin, there is perennial flow because of mine discharge. In Fig. 3.17 are the average monthly and annual discharges for the Rio San Jose and the Rio Puerco in comparison to cumulative annual flow from 14 mines, each discharging at 2 m^3/min. The monthly data reveal pronounced seasonal variations approaching 1 to 2 orders of magnitude. The streams do not show the same seasonal variations, further attesting to varied patterns of runoff, irrigation diversion, and control features such as impoundments and conveyance/ irrigation channels. Figure 3.18 shows the percentage of each month during
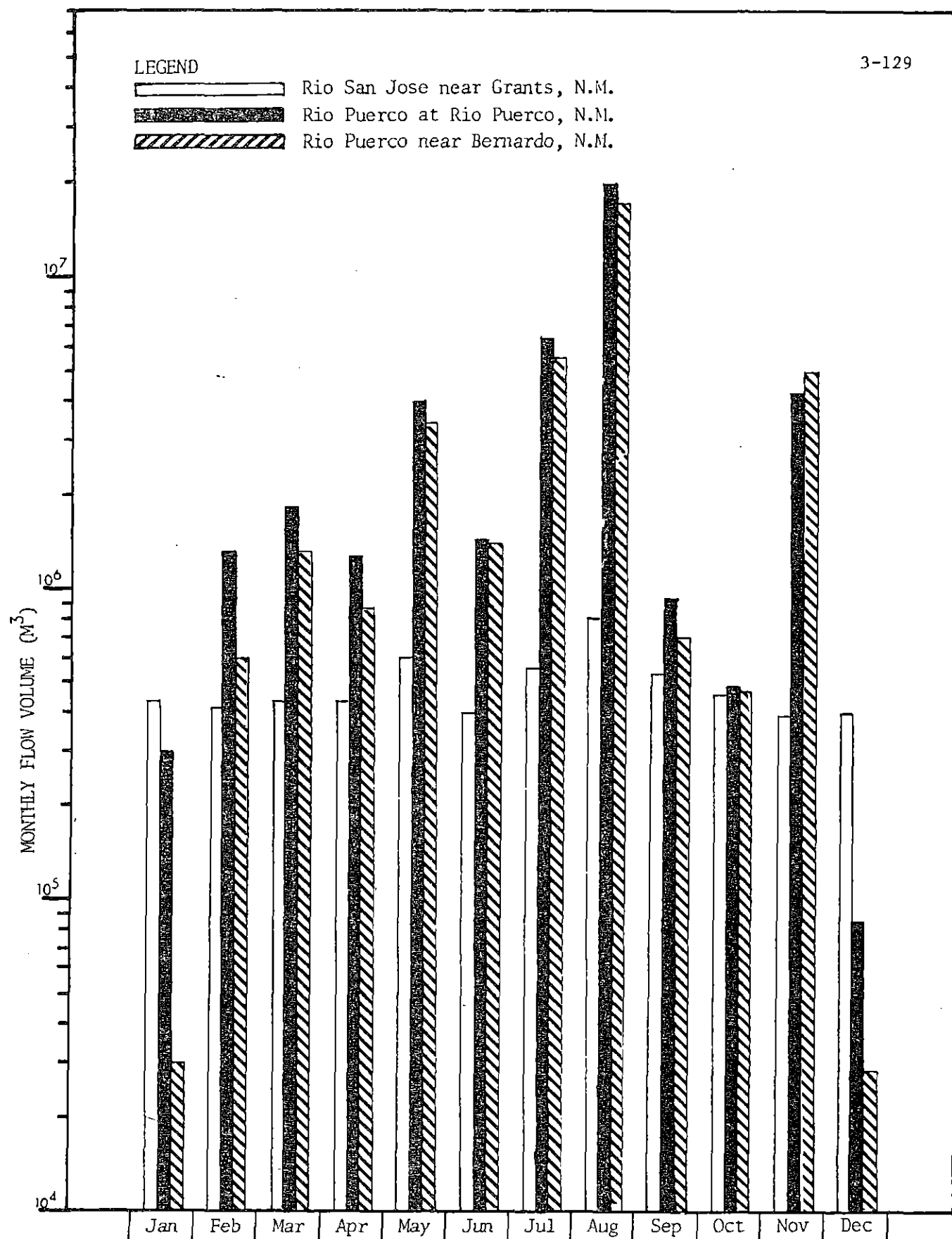
Figure 3.17 Average monthly flows for the period of record for the Rio San Jose and the Rio Puerco in New Mexico(Summarized from flow records provided by L. Beal, U.S. Geological Surve   erque).

which there is no flow in the Rio San Jose and Rio Puerco. The average period of annual or monthly no flow is as follows:

Rio San Jose near Grants: 0 Percent

Rio Puerco at Rio Puerco: 45 Percent

Rio Puerco at Bernardo : 71 Percent

It is also assumed that flow from the sub-basin reaches the first major stream, the Rio San Jose, with no change in flow or quality. Runoff is minimal in the lower reaches of San Mateo Creek because of internal drainage and considerable infiltration. Historical evidence indicates that only rarely, if ever, would flood runoff from Ambrosia Lake enter the Rio San Jose. In the interests of conservatism, total flow laden with contaminants is transported to the Rio San Jose. Dilution first occurs within the sub-basin and then, successively, in the Rio San Jose, Rio Puerco, and Rio Grande. The latter is the regional basin.

There is an infinite number of combinations of flood volumes and dilution volumes for the sub-basin, basin, and regional basin streams. Use of average annual discharge volumes in the receiving streams simplifies what would otherwise be a burdensome, confusing series of calculations. Flushing action from the sub-basin is handled on a probabilistic basis in terms of flow duration and return period. Concentration values are based on 14 mines, a loading period of two years, and flow and water quality data shown in Table 3.39. When, for example, 5-year or 10-year events are considered, it is conservatively assumed that events with shorter return periods do not occur. The accretion period remains constant (2 years), and only the return period and duration are varied, resulting in varying flow volumes. It is conceivable that contaminants could concentrate for 3, 4, or 5 years and then be flushed by a 2-year event, but this was not evaluated.

Minimum and maximum return periods for floods from the sub-basin were set at 2 and 25 years, respectively, for several reasons. The 2-year event, i.e., runoff volume over a duration of 1 day or 7 days and occurring on the average of every 2 years, is expected to occur rather frequently over the life of the mines (17 years). The intermediate-sized storms with return periods of 5 or 10 years would result in considerable contaminant transport,
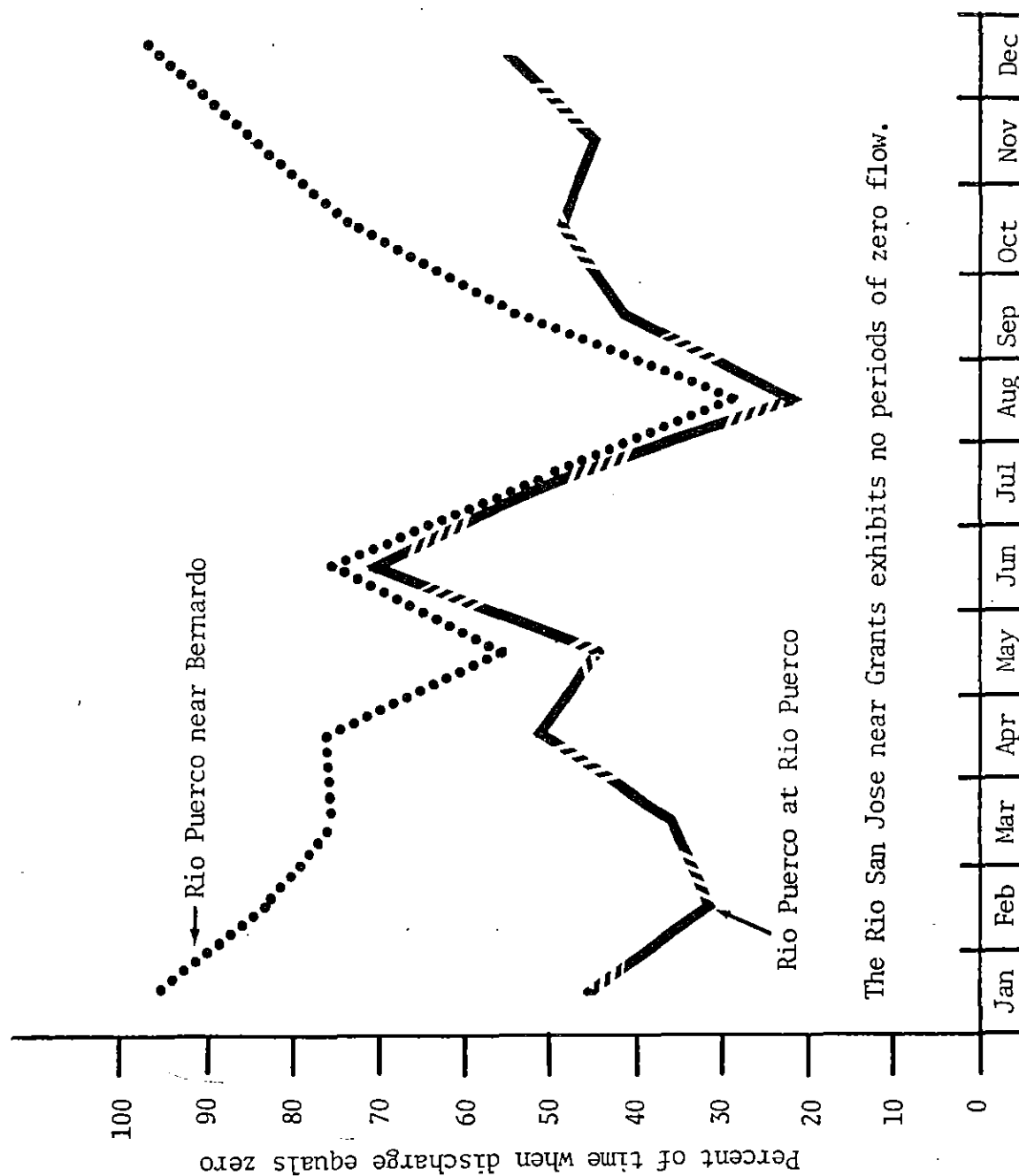
Figure 3.18 Periods of no flow in the Rio San Jose and Rio Puerco (Summarized from flow records provided by L. Beal, U.S. Geological Survey, Albuquerque).

but concentrations would be low owing to dilution and to annual or semiannual scouring provided by smaller floods. The 25-year event is a practical maximum expected to occur during the lifetime of the mining district. Still larger floods, with return periods of 50 or 100 years, can be calculated but are less important because of their infrequent occurrence. Figures 3.19 and 3.20 show calculated flow volumes from the sub-basin for 1-day and 7-day durations and return periods of 2 years to 50 years. The extreme range in flow volume is from 2.16×10^4 m^3 to 2.55×10^5 m^3.

Figures 3.19 and 3.20 show flow values in the Rio San Jose and Rio Puerco for 1-day and 7-day durations and return periods of 1 to 100 years. For the Rio San Jose, 1-day volumes range from 1.24×10^4 m^3 to 1.68×10^6 m^3. The mean annual discharge rate in the Rio San Jose is 11.09 m^3/min. Flow from the Rio San Jose enters the Rio Puerco where corresponding flows (1-day duration) range from 0.6×10^6 to 2.15×10^7 m^3 at the point of inflow to the Rio Grande. Average daily discharge in the Rio Grande seasonally ranges from 8.87×10^5 m^3 to 59.5×10^5 m^3. Average annual flows rather than peak 1-day or 7-day flows were used in the subsequent calculations.

The maximum probability for peak runoff from the sub-basin and resulting contaminant transport is in the summer months, at which time the Rio Puerco has no flow about 22 to 75 percent of the time. Flow in the Rio San Jose and Rio Puerco from June through September ranges from 3.96×10^5 to 1.97×10^7 m^3 per month for the period of record (Fig. 3.17).

3.4.3.1.4 Prediction of Sub-basin Water Quality

Table 3.43 outlines dilutions based on the foregoing discussion of flow patterns and discharges and considering only the 1-day sub-basin flood event with a 2-year recurrence interval. The dilution constant is the ratio of concentration in the receiving water to that in the contaminated (relatively) inflow. It is more commonly expressed as the dilution factor, which is the reciprocal. Thus, in the case of the sub-basin flood flow entering the mean annual flow of the Rio San Jose, there is a 271:1 dilution (Table 3.43).

With development of the foregoing (mine water) source term and surface water pathway, the remaining discussion emphasizes contaminant concentrations in surface water. This, in turn, serves as input data to health effects modeling for the water pathway. Chemical concentrations in the Rio
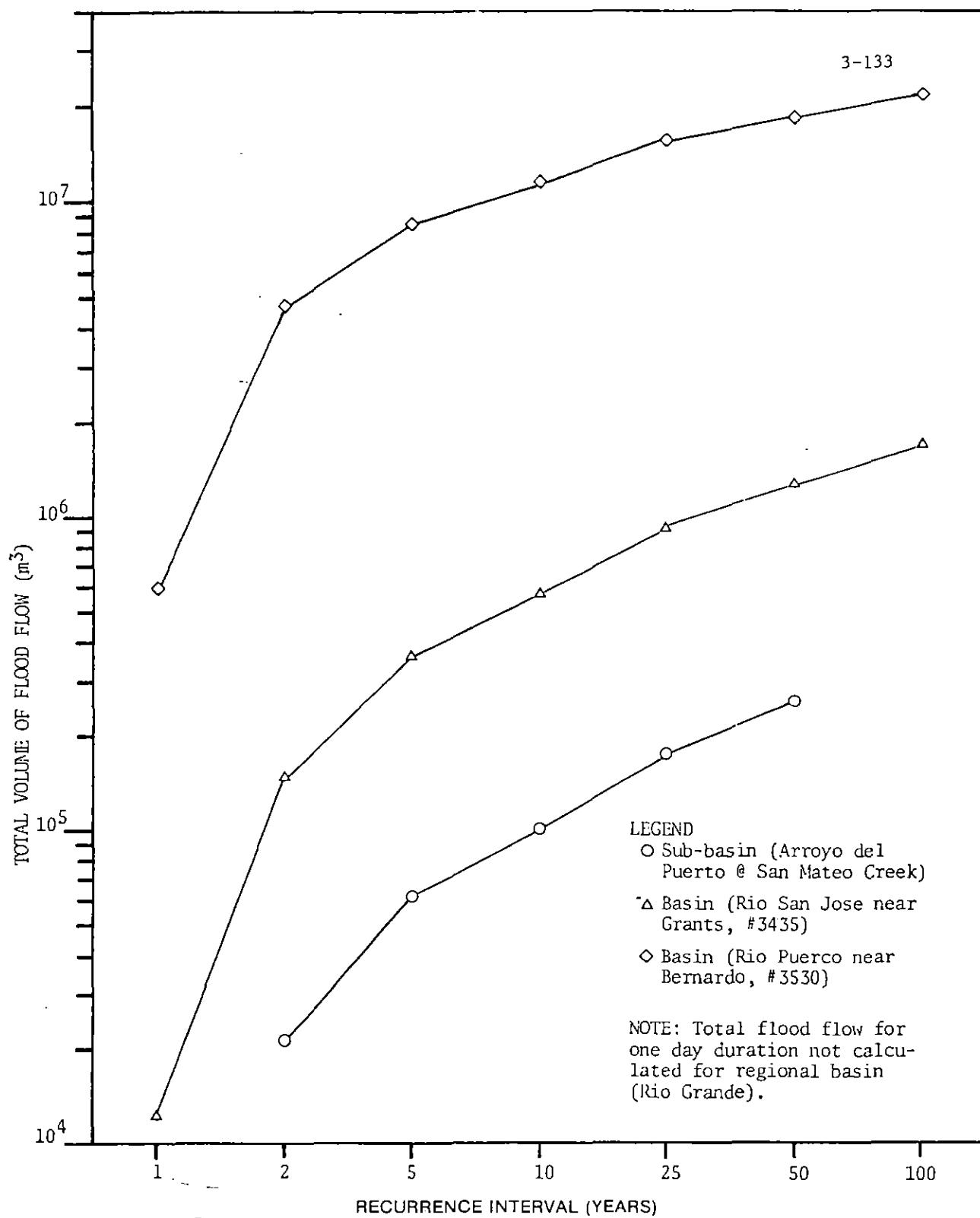
Figure 3.19 Total flow volumes in one-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico (Summarized from flow records provided by L. Beal, U.S. Geological Survey, Albuquerque).

LEGEND

o Sub-basin (Arroyo del Puerto @ San Mateo Creek)

△ Basin (Rio San Jose near Grants, #3435)

◇ Basin (Rio Puerco near Bernardo, #3530)

NOTE: Total flood for seven days duration not calculated for regional basin (Rio Grande).

TOTAL VOLUME OF FLOOD FLOW (m^3)

RECURRENCE INTERVAL (YEARS)
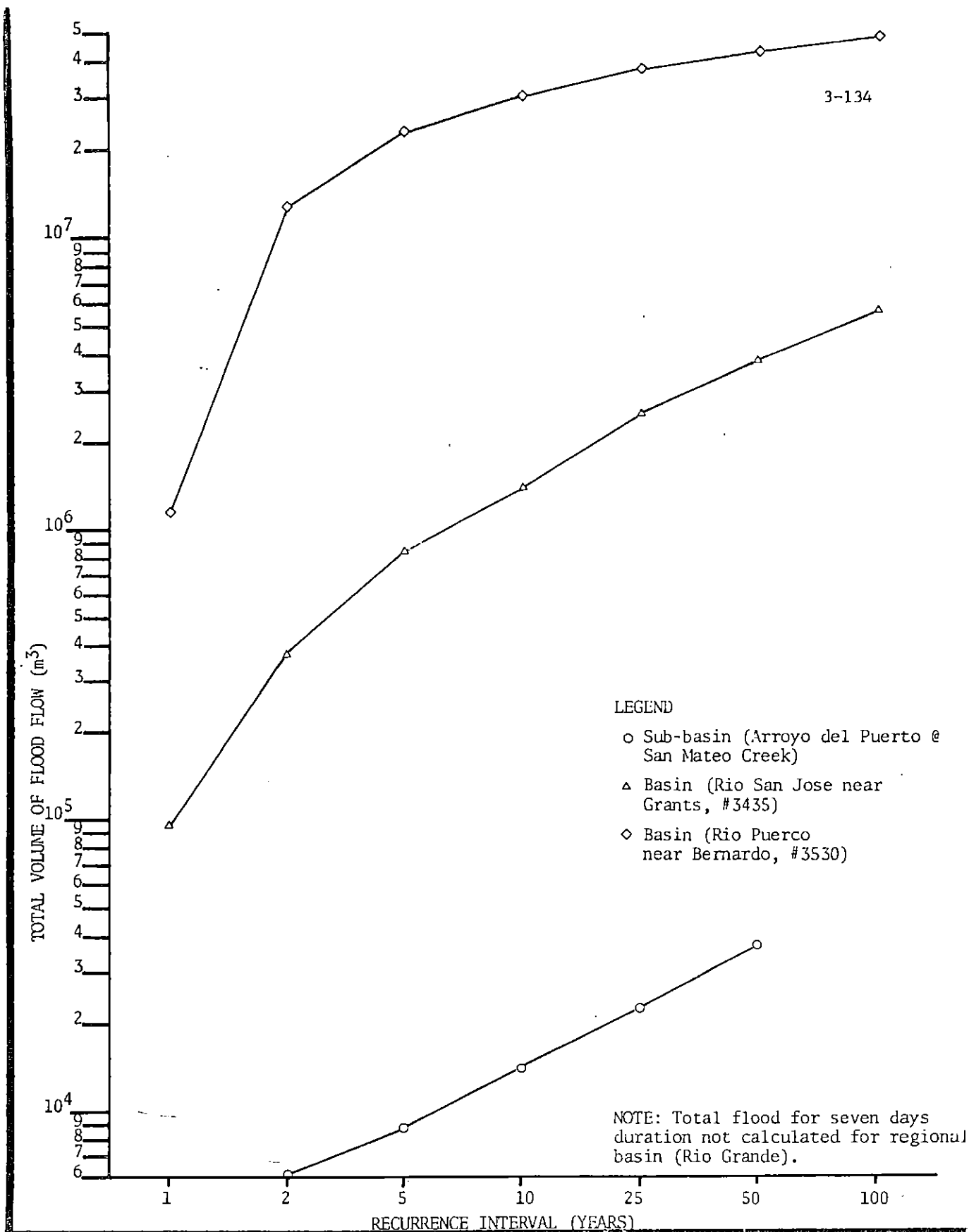
Figure 3.20 Total flow volumes in seven-day periods for floods of various recurrence intervals in the sub-basin and basins in New Mexico (Summarized

Table 3.43 Dilution factors for the Rio San Jose, Rio Puerco, and Rio Grande for 1-day flood flows with a 2-year recurrence interval

Hydrographic Basins	Flow Ratio (m^3/m^3)		Dilution Constant	Dilution Factor
Rio San Jose near Grants[a]	$\dfrac{2.16 \times 10^4}{5.83 \times 10^6 + 2.16 \times 10^4}$	=	0.0037	271
Rio Puerco[b]	$\dfrac{2.16 \times 10^4}{4.26 \times 10^7 + 2.16 \times 10^4}$	=	0.00051	1973
Rio Grande near Bernardo[c]	$\dfrac{2.16 \times 10^4}{86.69 \times 10^7 + 2.16 \times 10^4}$	=	0.000025	40135

[a]Calculated using mean annual flow in the Rio San Jose (near Grants, NM) station:

$$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio San Jose flow} + \text{Sub-basin flood flow}}$$

[b]Assumes Rio San Jose enters the Rio Puerco at Bernardo:

$$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio Puerco flow (includes Rio San Jose flow)} + \text{Sub-basin flood flow}}$$

[c]Dilution =

$$\text{Dilution} = \frac{\text{Sub-basin flood flow}}{\text{Rio Puerco flow} + \text{Rio Grande flow (at Bernardo)} + \text{Sub-basin flood flow}}$$

San Jose, Rio Puerco (at Bernardo), and in the Rio Grande (near Bernardo) are shown in Table 3.44 along with 1-day and 7-day flood flow volumes from the sub-basin for return periods of 2, 5, 10, and 25 years. These flood volumes are diluted into the mean annual flow of the Rio San Jose (near Grants), Rio Puerco (at Bernardo), and Rio Grande (near Bernardo). The principal reason for using mean annual flow is that the radiation dose and health effects model (Section 6.0) stresses estimating average annual dose to the population over the duration of mining activity.

For example, the 1-day duration flood flow (with a 2-year return period) contains 1920 mg/ℓ uranium, which decreases to 7.09 mg/ℓ in the Rio San Jose and 0.973 mg/ℓ in the Rio Puerco. Because of the short duration of most floods in the sub-basin, there is little difference in flow volume and, thus, dilution between the 1-day and 7-day events. With progressive dilution down-stream, the difference in size between sub-basin floods of varying durations and return periods becomes insignificant relative to the mean annual flow volumes of the basin and regional basin streams. As a result, concentrations tend to reach a minimum and remain unchanged at this degree of accuracy.

As in the case of the Wyoming surface-mine scenario, we assume that most contaminants in the mine water collect on or near the land surface and are available for transport. This assumption is open to ques-tion, but field data are scarce to support contentions as to the fraction of contaminant load that becomes unavailable. For example, extensive field studies along the Animas, San Miguel, and Dolores Rivers in Colorado con-cluded that "...once radium becomes a part of a stream's environment, it constitutes a relatively long-term and continuous source of water and aquatic biota contamination" (Si66). However, cessation of uranium mill discharges to the Colorado River tributaries effectively negated this source, which is now believed to be buried behind the Lake Powell and Lake Mead impoundments. Similarly, dissolved radium reverts to background levels of several pico-curies per liter in natural streams receiving mine water in Colorado and New Mexico. Although it is likely that flood waters resuspend precipitates and sediments with sorbed radium, laboratory experiments (Sh64; Ha68) indicate that only minor re-solution takes place. This phenomenon is supported by recent surface water data collected in the Grants Mineral Belt of New Mexico (Ku79). Therefore, concentrations of dissolved radium in flood water are

Table 3.44 Annual contaminant loading from 14 uranium mines and resulting concentrations in sub-basin floods and in the average annual flow of the Rio San Jose, Rio Puerco, and Rio Grande

Contaminant concentration in mine effluent (mg/ℓ except as noted)	Mass available for transport (kg/yr except as noted)[a]	1- and 7-day flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 25 years[b]							
		1-Day				7-Day			
		$V_2=2.16\times10^4$ C_2	$V_5=6.23\times10^4$ C_5	$V_{10}=1.02\times10^5$ C_{10}	$V_{25}=1.76\times10^5$ C_{25}	$V_2=5947$ C_2	$V_5=8794$ C_5	$V_{10}=1.43\times10^4$ C_{10}	$V_{25}=2.26\times10^4$ C_{25}
Total Uranium 1.41	1480	1920 7.09 0.973 0.0456	665 7.03 0.971 0.0456	406 6.98 0.970 0.0455	235 6.89 0.967 0.0455	6970 7.10 0.973 0.0456	4710 7.09 0.972 0.0455	2900 7.10 0.973 0.0456	1830 7.07 0.970 0.0456
Radium-226 13.7 pCi/ℓ	0.00144 Ci/yr	1870 6.9 0.95 0.044	647 6.8 0.95 0.044	395 6.8 0.94 0.044	229 6.7 0.94 0.044	6780 6.9 0.95 0.044	4580 6.9 0.95 0.044	2820 6.9 0.95 0.044	1780 6.9 0.94 0.044
Lead-210 14.6 pCi/ℓ	0.00153 Ci/yr	19800 73.1 10.0 0.470	6880 72.7 10.0 0.471	4200 72.2 10.0 0.471	2430 71.2 10.0 0.470	72000 73.4 10.0 0.471	48700 73.3 10.1 0.471	30000 73.4 10.1 0.472	19000 73.4 10.1 0.472
Cadmium 0.007	7	9 0.03 0.004 0.0002	3 0.03 0.004 0.0002	2 0.03 0.005 0.0002	1 0.03 0.004 0.0002	30 0.03 0.004 0.0002	20 0.03 0.004 0.0002	10 0.02 0.003 0.0002	9 0.03 0.005 0.0002
Arsenic 0.012	13	17 0.063 0.0086 0.00040	5.8 0.061 0.0085 0.00040	3.6 0.062 0.0086 0.00040	2.1 0.062 0.0086 0.00041	61 0.062 0.0085 0.00040	41 0.062 0.0085 0.00040	25 0.061 0.0084 0.00039	16 0.062 0.0085 0.00040
Selenium 0.076	80	100 0.38 0.053 0.0026	36 0.38 0.052 0.0026	22 0.38 0.052 0.0026	13 0.37 0.052 0.0026	376 0.38 0.053 0.0026	254 0.38 0.053 0.0026	156 0.38 0.053 0.0026	99 0.38 0.053 0.0026

Table 3.44 (continued)

Contaminant concentration in mine effluent (mg/ℓ except as noted)	Mass available for transport (kg/yr except as noted)[a]	1-Day $V_2=2.16\times10^4$ C_2	1-Day $V_5=6.23\times10^4$ C_5	1-Day $V_{10}=1.02\times10^5$ C_{10}	1-Day $V_{25}=1.76\times10^5$ C_{25}	7-Day $V_2=5947$ C_2	7-Day $V_5=8794$ C_5	7-Day $V_{10}=1.43\times10^4$ C_{10}	7-Day $V_{25}=2.26\times10^4$ C_{25}
Molybdenum 0.29	300	390	130	82	48	1400	960	590	370
		1.4	1.4	1.4	1.4	1.4	1.4	1.4	1.4
		0.20	0.19	0.20	0.20	0.20	0.20	0.20	0.20
		0.0093	0.0089	0.0092	0.0091	0.0092	0.0093	0.0091	0.0092
Barium 0.81	850	1100	380	230	140	4000	2700	1700	1100
		4.1	4.0	4.0	4.1	4.1	4.1	4.2	4.2
		0.56	0.55	0.55	0.58	0.56	0.56	0.57	0.58
		0.026	0.026	0.026	0.027	0.026	0.026	0.027	0.027
Zinc 0.043	45	58	20	12	7.2	210	140	88	56
		0.21	0.21	0.21	0.21	0.21	0.21	0.22	0.22
		0.029	0.029	0.029	0.030	0.029	0.029	0.030	0.030
		0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014	0.0014
Sulfate 580	1.22×10^5	1.58×10^5	5.48×10^4	3.35×10^4	1.94×10^4	5.74×10^5	3.88×10^5	2.38×10^5	1.51×10^5
		584	580	574	568	586	584	582	584
		80	80	80	80	80	80	80	80
		3.8	3.8	3.7	3.8	3.8	3.8	3.7	3.8
Total Suspended Solids 27.8	29000	38000	13000	8000	4600	140000	92000	57000	36000
		140	140	140	130	140	140	140	140
		19	19	19	19	20	19	19	19
		0.90	0.89	0.90	0.89	0.92	0.89	0.90	0.89

Header note: 1- and 7-day flood flow volumes (m^3) and contaminant concentrations associated with return periods of 2 to 25 years[b]

(a) Mass values shown are on an annual, per-mine basis.

(b) V_r and C_r refer respectively to flood volume, in cubic meters, and concentration in runoff for an r-year flood. Concentrations are in mg/ℓ, except for radium-226 and lead-210, which are in pCi/ℓ. Concentrations shown are from accretion or loading in the sub-basin for 2, 5, 10, 25 years, yielding the first value shown in each set. The next three values below this initial value represent, in downward order, concentrations in the flood flow as diluted by the mean annual flow in 1) the Rio San Jose near Grants (5.83×10^6 m^3), 2) the Rio Puerco at Bernardo (4.26×10^7 m^3), and 3) the Rio Grande near Bernardo (86.69×10^7 m^3).

Note.--Assumptions: Mines discharge continuously at a rate of 2.0 m^3/min. Concentrations are the average of those shown in Table 3.39. Except for radium and sulfate, all suspended and dissolved contaminants remain in or on the stream sediments and are mobilized by flood flow. Twenty percent of the sulfate and 10 percent of the radium are available for resolution.

arbitrarily set at 0.00144 Ci/yr or 10 percent of the annual loading from the model mine.

Sulfate is also considered an important exception in the total "transport" concept. Because sulfate can be a highly mobile anion, it is assumed that 80 percent of the load enters the shallow groundwater reservoirs and 20 percent is available for solubilization and chemical transport in surface flows. No distinct pattern of groundwater contamination from mine water, per se, was documented in an earlier Grants Mineral Belt survey (EPA75), but recent data from the State indicate groundwater deterioration as a result of mine drainage (J. Dudley, New Mexico Environmental Improvement Division, oral communication, 1979). It is likely that considerable fractionation of other stable and radioactive trace elements occurs, but field data specific to the uranium mining regions are quite scarce, with the exception of Texas (He79), where only stable elements were studied. Because of our imperfect, non-predictive understanding of trace element transport in aqueous systems, our analysis assumes total transport for most constituents in lieu of numerous, equally unfounded assumptions for resuspension factors, fractionation, etc. Floods of 1-day and 7-day duration and return periods of 2, 5, 10, and 25 years are arbitrarily selected as providing the necessary flushing action associated with intense, short-term runoff events. It is likely that storms of shorter (less than 1-day) duration and possibly greater discharge rate also transport contaminants. The flow volume and thus the dilution cannot be estimated for these events.

Calculated water quality in basin and regional basin streams is shown in Table 3.45 along with established and suggested standards for selected contaminants. For uranium, concentrations in the basin exceed the suggested limits based on chemical toxicity and radiotoxicity. Radium-226/228 exceeds the standard in the basin but is well below the standard for the regional basin. The same is true for sulfate, cadmium, arsenic, barium, and selenium. Zinc is the only contaminant consistently below the potable and irrigation water standards. As in the case of the surface mine scenario for Wyoming, uranium is apparently well above suggested limits and warrants further study, as do the stable toxic elements in the basin area(s) closest to the mining centers.

With the exception of radium-226 and sulfate, the concentrations of radionuclides and other parameters shown in Tables 3.44 and 3.45 reflect no

Table 3.45   Comparison of potable and irrigation water standards and surface water quality affected by underground mine drainage

Parameter	Range of contaminant concentrations in flood flow affected by mine discharge[a]				Potable water standards (mg/ℓ)[b]		Irrigation[c] Recommendations for maximum concentration for continuous use on all soils (mg/ℓ)
	Basin Min.	Basin Max.	Regional Basin Min.	Regional Basin Max.	Maximum Permissable Concentration	Recommended Limiting Concentration	
Total U	6.9	7.1	0.045	0.046	0.015/3.5/0.2[d]		---
Ra-226 + 228	6.7	6.9	0.044	0.044	----	5 pCi/ℓ	5 pCi/ℓ
TSS	130	140	0.89	0.92	--	--	--
Sulfate	574	584	3.7	3.8	--	250	200
Zn	0.21	0.22	0.0014	0.0014	--	5.0	2.0
Cd	0.03	0.03	0.0002	0.0002	0.01	--	0.010
As	0.061	0.063	0.00039	0.00041	0.05	0.01	0.10
Ba	4.0	4.2	0.026	0.027	1	--	---
Se	0.37	0.38	0.0026	0.0026	0.01	--	0.02

[a]Concentrations in milligrams per liter, except Ra-226 -228 which are in picocuries per liter. Data shown apply to the Basin (Rio San Jose near Grants) and Regional Basin (Rio Grande near Bernardo) streams (Table 3.44).

[b]Sources: U.S. Environmental Protection Agency (EPA76) and, in the case of uranium, suggested guidance from the National Academy of Sciences (NAS79) to the USEPA and from USEPA, (Office of Drinking Water) to the State of Colorado (La79).

[c]Source: (NAS72).

[d]0.015 mg/ℓ :Suggested maximum daily limit based on radiotoxicity for potable water consumed at a rate of 2 liters per day on a continuous basis
3.5 mg/ℓ: Suggested maximum daily limit based on chemical toxicity and intake of 2 liters in any one day
0.21 mg/ℓ: Suggested maximum daily limit based on chemical toxicity and intake of 2 liters per day for 7 days

reductions for ion exchange, precipitation, or sorption. Rather, a simple dilution model is used in which the mass loading from mine discharge is calculated as the product of concentration and discharge (volume). There are problems with this approach. In some cases, the calculated concentrations in flood waters probably exceed the solubility limits, as in the case of sulfate in the presence of barium. In other instances, precipitation of barium sulfate or iron and manganese hydroxides might greatly reduce the concentration of radium and uranium, both of which would coprecipitate. Thus the stream concentrations shown in Table 3.44 are probably high (conservative). To improve the analysis, additional comparisons or parallels were drawn using mill tailings solutions and stream water quality as affected by mine drainage and a mill tailings spill.

Contaminant concentrations in uranium mill tailings liquids provide an upper limit estimate of runoff concentrations insofar as the solvent action of tailings solutions maximize dissolution of minerals present in the ore (J. Kunkler, USGS, written communication, 1979). Table 3.46 is a compilation of mill tailings water quality data from numerous previous reports and summarized by EPA and USGS staff (Ka79; Ku79). It is apparent that there are wide variations as a function of mining region and whether an acid or alkaline leach mill circuit is used. The Nuclear Regulatory Commission (NRC79b) assumption for the composition of a "typical" acid leach mill is shown along with other average or representative analyses. A conservative (worst quality) analysis for uranium mill pond water quality is estimated as follows (Table 3.47) and compared to the average concentrations calculated from the mixing of mine effluent and flood volumes (Table 3.44).

The data in Table 3.47 suggest that calculated concentrations in the sub-basin almost without exception exceed those in uranium mill tailings solutions. Thus, the calculated values are probably erroneously high. Calculated concentrations in flood waters of the basin and regional basin streams are considerably less and are in rough agreement with field data, at least for the stable constituents. Radium-226 and lead-210, however, still seem excessively high considering the various natural processes of sorption, precipitation, and so on. To understand the degree to which natural streams transport contaminants, we reviewed water quality data from selected New Mexico streams receiving mine drainage.

Table 3.46 Radiochemical & stable element/compound water quality for selected acid & alkaline leach uranium mill tailings ponds in the United States

Tailings Pile Location	U (mg/ℓ)	Th-230	Ra-226 (pCi/ℓ)	Pb-210	Po-210	As	Mn	Cu	Se	Mo	V (mg/ℓ)	SO	Na	Fe	TDS	NH	Ca	NO	Cl
1. Split Rock, WY (acid)	10.5	41600	4800	---	940	1.1	15.5		0.2	1	0.05			280	11810	374	560	43.5	65
2. Canon City, CO (acid)	---	---	---	---	---	10.1	25.0	18	0.6	190	7.1	34000	19000	280	77400	---	380	140	6500
3. Moab, UT	2.0	50	100	---	---	7.0	---	--	---	3	---	100	---	---	150000	---	---	---	300
4. United Nuclear, NM (acid)(a)	14	---	38	---	---	---	50	3	0.005	3	30	---	300	1000	---	---	700	---	---
5. Anaconda Inj. Well Feed, NM	130	---	53	---	---	---	340	--	0.03	---	6.3	4900	1200	---	---	69	---	7.4	---
6. Kerr-McGee, NM (acid)(a)	32	---	58	---	---	---	30	5	0.18	7	10	---	500	1000	---	---	300	---	---
7. UN-HP, Grants, NM (alkaline)	150	---	52	---	---	---	---	--	0.92	70	6.8	4300	4300	---	---	4.4	---	4.4	2
8. Humeca, WY (acid)	68.4	110	240	---	---	---	0.4	0.1	---	---	---	6500	11700	0.5	---	460	---	16	16000
9. USNRC-Uranium Milling EIS(acid)	8.0	150000	400	400	400	0.2	500	50	20	100	0.1	30000	500	1000	35000	500	500	---	300
10. Representative acid millpond(a) in New Mexico	---	---	---	---	---	---	---	--	---	---	---	44000	---	170	---	300(b)	---	---	1800
"Average" (Exclusive of 9 and 10)	58	13920	760	---	---	6	160	6.5	0.32	54	10	10000	6200	510	80000	227	485	40	1700
Maximum value: "Average" versus NRC GEIS	58	150000	760	400	400	6	500	50	20	100	10	30000	6200	1000	80000	500	500	40	1700

(a) Source: Ku79.
(b) Ammonium ion.

Table 3.47   Summary of flood runoff water quality and
uranium millpond quality

Parameters	Concentration in uranium mill tailings solution	Concentration in flood waters of the sub-basin[a]	Concentration in flood waters of the Rio San Jose[a]
Uranium (mg/ℓ)	58	235 - 6970	7
Radium-226 (pCi/ℓ)	760	229 - 6780	6.9
Lead-210 (pCi/ℓ)	400	2430 - 72000	73
Polonium-210 (pCi/ℓ)	400	NC[b]	NC
Arsenic (mg/ℓ)	6	2.1 - 61	0.062
Manganese (mg/ℓ)	500	NC	NC
Copper (mg/ℓ)	50	NC	NC
Selenium (mg/ℓ)	20	11 - 220	0.34
Molybdenum (mg/ℓ)	100	48 - 2400	1.4
Vanadium (mg/ℓ)	10	NC	NC
Sulfate (mg/ℓ)	30,000	$9.7 \times 10^4 - 2.87 \times 10^6$	2901

(a) Refer to Table 3.44.
(b) Not calculated.

The USGS, by water sampling in the Churchrock area of New Mexico (J.L. Kunkler, USGS, written communication, 1979), determined water quality in an ephemeral stream receiving rather large and continuous mine discharges. Data are also available from the Schwarzwalder Mine near Golden, Colorado (EPA72). Until 1972, this mine discharged effluent high in uranium, radium, and trace elements to Ralston Creek and subsequently to two lakes/reservoirs used for irrigation and potable supply (Section 3.2.3.2.1).

The way in which surface runoff water quality is created or affected by mine discharge is complex. In the Churchrock area, numerous water quality changes occur as the mine discharges flow toward Gallup (Fig. 3.21 and Table 3.48). As in other uranium mining areas in New Mexico, stream volume constantly decreases with flow distance, but water quality changes are erratic. Infiltration, discussed in more detail in the following section of the report and in Appendix H, amounts to about 90 percent or more of the water loss. The balance is by evaporation. On a percentage basis, similar losses occur in the principal drainage courses in Ambrosia Lake. Dissolved Ra-226 decreases from 30 to 0.88 pCi/ℓ in a reach of 9.2 km and, on a later date, from 14 to 0.95 pCi/ℓ in a distance of 26.7 km. Based on the limited flow and water quality data, it appears that radium is strongly sorbed onto the stream sediments. In October 1975, soluble uranium decreased from 1150 to 740 μg/ℓ in the reach immediately below the mine discharges, yet in July 1977 and May 1978 uranium increased in the downstream direction from 580 to 860 μg/ℓ and from 970 to 2800 μg/ℓ. These changes bear no consistent relation to fluctuations in dissolved or suspended solids along the flow path. Both of the latter parameters appear to increase in the direction of flow and may be a result of flash floods in lower reaches of the basin. Uranium appears to undergo little change and may actually increase in the downstream direction. Of the stable trace elements, vanadium, selenium, iron, molybdenum, and zinc show no consistent change with distance.

A third approach used to assess surface runoff quality involved a brief review of some of the data collected to monitor a July 1979 tailings accident in New Mexico. The mill tailings dam at the Churchrock mill breached and dumped 223,000 m^3 of liquid and 1,000 metric tons of solids into the Rio Puerco drainage system. The catastrophe immediately spurred numerous water quality studies by State and Federal agencies. Numerous inter-
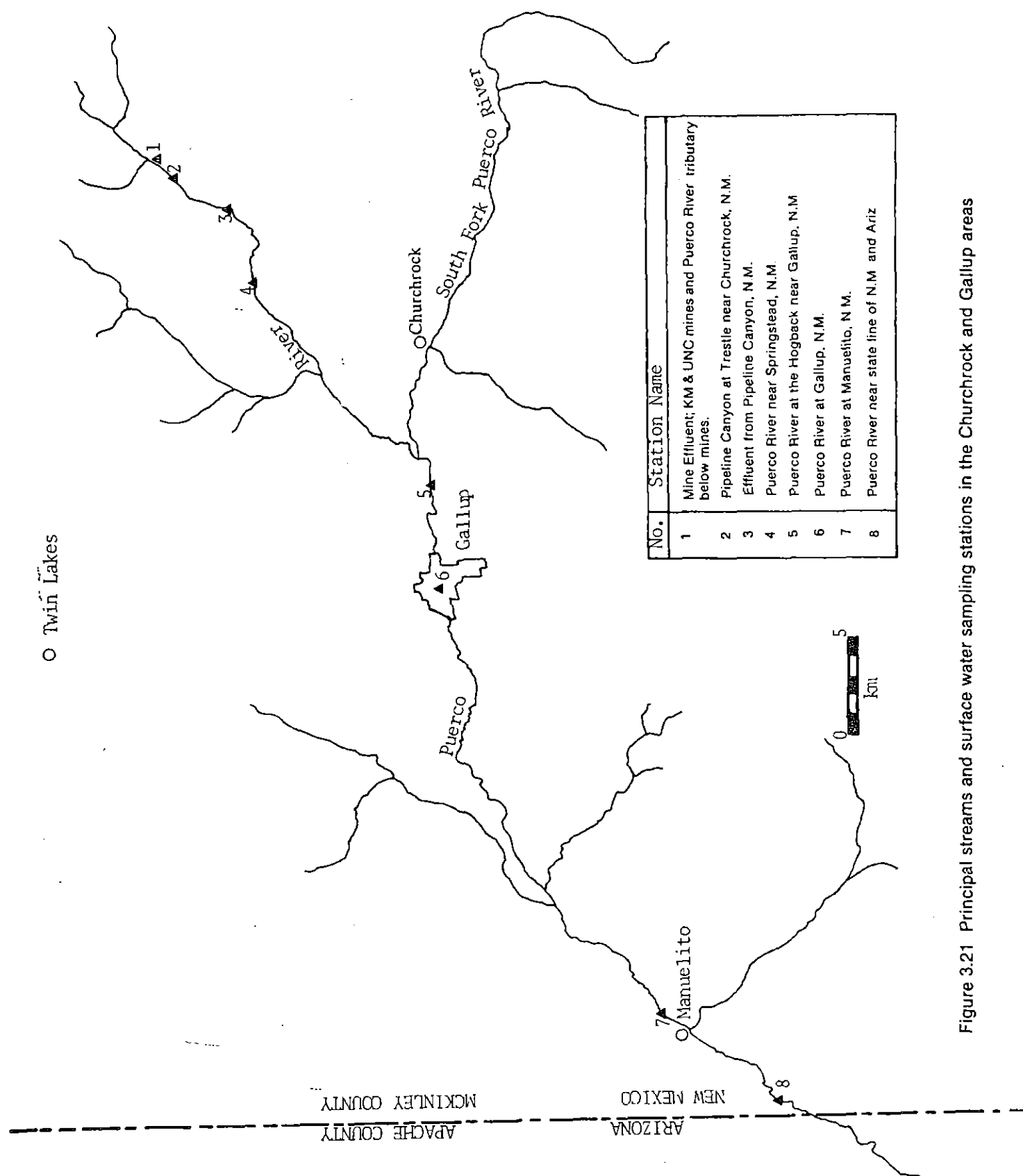
Figure 3.21 Principal streams and surface water sampling stations in the Churchrock and Gallup areas

No.	Station Name
1	Mine Effluent; KM & UNC mines and Puerco River tributary below mines.
2	Pipeline Canyon at Trestle near Churchrock, N.M.
3	Effluent from Pipeline Canyon, N.M.
4	Puerco River near Springstead, N.M
5	Puerco River at the Hogback near Gallup. N.M
6	Puerco River at Gallup. N.M.
7	Puerco River at Manuelito, N.M.
8	Puerco River near state line of N.M and Ariz

Table 3.48   Flow and water quality in the Puerco River near Churchrock and Gallup, New Mexico

Location and (Station Number)	m^3/min	U nat. μg/ℓ as U_3O_8	Ra-226 pCi/ℓ	Total Solids, mg/ℓ Dissolved	Total Solids, mg/ℓ Suspended	Suspended solids, metric tons per day	Concentrations μg/ℓ Ba	Cd	Cr	Pb	Mo	V	Zn	Se	As	Fe
Oct. 16, 1975																
Puerco River tributary below mines - (1)	14.5	1150	30	430	410	9.5	-	-	-	-	-	21	-	27	-	20
Puerco River near Springstead, NM -(4)	12.4	740	0.88	480	1600	8.67	-	-	-	-	-	13	-	25	-	30
Puerco River at Gallup, NM - (6)	5.11	---	0.52	640	2300	5.14	-	-	-	-	-	5.7	-	26	-	40
Puerco River at Manuelito - (7)	6.8(est)	540	0.25	800	2800	---	-	-	-	-	-	-	-	-	-	-
July 6, 1977																
Puerco River tributary below mines - (1)	11.55	580	14	410·	260	---	800	1	0	6	-	-	0	25	1(3)	10
Puerco River at the Hogback, near Gallup, NM - (5)	6.47	860	0.95	520	15000	---	100	1	0	11	-	-	50	20	1(19)	80
Puerco River near State line (NM/AZ) - (8)	15.5	83	0.27	600	44000	---	1700	4	0	2	-	-	30	5	6(7)	90

Table 3.48 (Continued)

Location and (Station Number)	m³/min	U nat. as U_3O_8 μg/ℓ	Ra-226 pCi/ℓ	Total Solids, mg/ℓ		Suspended solids, metric tons per day	Concentrations μg/ℓ									
				Dissolved	Suspended		Ba	Cd	Cr	Pb	Mo	V	Zn	Se	As	Fe
May 25, 1978																
Effluent from Kerr McGee and United Nuclear Mines, Churchrock, NM- (1)	10.9	807	2.6	---	---	---	-	-	-	-	12	19	-	4	-	-
Effluent from Pipeline Canyon, NM - (3)	9	2800	1.5	---	---	---	:	-	-	-	820	28	-	110	-	-
Puerco River near Springstead, NM - (4) (sampled 5/18/78)	10.88	1100	0.8	---	---	---	-	-	-	-	12	16	-	0	-	15
July 11-12, 1978																
Pipeline Canyon at trestle near Churchrock, NM - (2)	14.45	940	8.6	---	---	---	-	-	-	-	230	11	-	-	-	540
Effluent from Pipeline Canyon, NM - (3)	14.3	1120	1.3	---	---	---	-	-	-	-	260	6	-	11	-	70
Puerco River near Springstead, NM - (4)	---	1130	2.2	---	---	---	-	-	-	-	240	9	-	13	-	40

Source: New Mexico District office of the U.S. Geological Survey (Peter Frenzel, written communication, 1979 and Kunkler, 1979).

pretations of the data have led to some confusion, compounded in some instances by inconsistent sample collection and preservation. However, several general findings seem true. Dissolution of stable and radioactive trace contaminants in flood waters does not seem significant providing that pH of the flood is in the range of 4 to 7. After several days, the mill tailings liquid was diluted and neutralized and contaminant concentrations decreased -- sometimes to levels lower than before the accident (J. Kunkler, USGS, written communication, 1979). At a downstream sampling station near Gallup, some 30 kilometers from the spill, dissolved uranium and radium-226 about 36 hours after the spill were 3.1 mg/ℓ and 0.95 pCi/ℓ, respectively. Suspended sediments contained 19 ppm uranium and 0.72 pCi/g radium-226. For the latter, this is less than background.

The surface water quality data pertaining to discharge of mine effluents and to the July 1979 spill seem to indicate rapid and thorough removal of radium-226 as a result of sorption, precipitation, pH adjustment, etc. However, stream sediment analyses in the Grants Mineral Belt are scarce, and there are no analyses of suspended solids in flood waters. Stream-bed sediment analyses by the USGS indicate less sorbed radium-226 and uranium than expected (Ku79). During this spill incident, uranium and selenium were relatively mobile in surface streams.

From the foregoing review of the literature and field data and preliminary calculations of runoff quality (Table 3.44), the following general conclusions are offered:

1. Radium-226 is removed from surface water in the New Mexico study area at rates of 0.5 to 3 pCi/ℓ per kilometer of stream. Final concentrations are on the order of 0.25 pCi/ℓ. Resolution in successive surface flows occurs, but it is not significant.

2. Uranium and certain stable trace elements, such as selenium, vanadium, molybdenum, and iron, show no consistent reduction with flow distance and may show an increase, at times.

3. Considerable more data collection is needed to understand the fate of dissolved and suspended contaminants from mine drainage. The present data base is rather limited in terms of sampling frequency, variety of contaminants measured, and types of measurements, for example, suspended solids analyses for flood waters.

4. With the exception of radium-226, the preliminary calculations of runoff quality in Table 3.44 are believed to be a first approximation of field conditions. Additional studies specific to the principal mining districts are needed.

5. Dissolved radium-226 concentrations in runoff are believed to be several picocuries per liter or less under natural conditions.

6. Uranium is fairly mobile and probably the most significant radionuclide in uranium mine effluent.

3.4.3.2 Impacts of Seepage on Groundwater

The principal use of groundwater in the immediate area of the mines is for stock water. Wells in the highland areas are typically one to two hundred meters deep and completed in underlying bedrock strata (Co68; Ka75). Contamination of such wells by mine discharge is considered extremely unlikely. Shallow wells are few in number and located along major drainages that are typically ephemeral. Such shallow wells are susceptible to contamination if located downgrade from mine discharges. Municipal water supplies are usually developed from wells because groundwater is consistently available and has acceptable suspended and dissolved mineral contents. The aquifers tapped by municipal wells are mostly either quaternary lava flows or deeper mesozoic sandstone and carbonate sequences. Considering the distance from the mining centers to the communities and the hydrogeologic conditions, it is unlikely that mining will cause measurable deterioration of municipal water quality. The greatest likelihood for contaminated groundwater is in the shallow, alluvial aquifer beneath streams receiving mine drainage. It is extremely unlikely that water quality in deeper, artesian aquifers will be adversely affected by mine discharge or overland flow affected by solid wastes. Shallow wells in these locations have been constructed in the past, but there are only a few and they are used for stock watering. It is possible that recharge of substantial quantities of mine water to the shallow aquifer will encourage additional use of it, in which case water quality will be of concern.

Table 3.49 shows average and extreme concentrations of various common and trace constituents in groundwater and other measures of water quality. The data are composited from a previous study (EPA75) and from unpublished

analyses by the New Mexico Environmental Improvement Division (J. Lazarus, NMEID, oral communication, 1979). We have categorized the data according to principal aquifers, which are in areas where the groundwater is not believed to be contaminated by mining. Because it is common for a well to tap more than one aquifer, the differences in water quality in Table 3.49 are approximate at best. The data reveal no sharp differences in water quality amongst the three major aquifers. The San Andres Limestone, a major aquifer for municipal and industrial uses in the Grants and Milan areas, has equal or greater concentrations of most constituents as compared to the Westwater Canyon Member and Gallup Sandstone units, which are closely associated with uranium mineralization.

Theoretical analysis of radionuclide transport in groundwater beneath and adjacent to a uranium mill tailings pond reveals very limited migration of radionuclides in groundwater (Se75). Using a seepage rate of 4×10^{-7} cm/sec and a 10 percent loss of soluble radionuclides, numerical solutions for steady state flow and transport into unconsolidated sand for periods of 5 years and 20 years reveal up to several meters movement of radium-226, thorium-230/ 234, uranium, and lead-210 after 20 years of leaching. For example, radium in groundwater to a depth of 3 meters is 10 percent of that in the tailings pond. Because the other isotopes tend to have even greater sorption, migration distances are further reduced. Although field studies at three uranium mill tailings piles in the Grants Mineral Belt substantiate only local migration of radionuclides (EPA75), extensive lateral migration of stable chemical species has been observed at uranium mills in Colorado, Wyoming, and Washington (Ka79, Ka78a, He79). For example, with respect to the old Cotter uranium mine at Canon City, Colorado, the Colorado Department of Health has stated in its Final Executive Licensing Summary, August 17, 1979, that "contamination attributed to tailings liquid was observed in an off-site water well ten years after the mill began depositing tailings, a [migration] rate of over five hundred feet per year." With respect to the same site, one researcher has stated that "the soluble uranium content of Lincoln Park ground waters is highly elevated with respect to Arkansas River water and exceeds suggested thresholds below which ecological and health effects are not expected. Molybdenum concentrations in these ground waters greatly exceed irrigation standards as well as the ALG based on health and ecological effects...." (Dr79). Near neutral pH and relatively low concen-

Table 3.49 Groundwater quality in principal aquifers in the
Grants Mineral Belt, New Mexico

		Aquifer	
Parameter	San Andres Limestone	Westwater Canyon Member, Morrison Fm. and Gallup Sandstone	Quaternary Alluvium, Tertiary Volcanics, and Chinle Formation
pH	7.2[a] (6.9 - 7.5)	7.9 (6.7 - 9.15)	7.6 (6.25 - 8.8)
Spec. cond. μmhos/cm	1900 (720 - 3500)	1800 (550 - 4250)	1715 (700 - 4000)
TDS	1680 (490 - 4500)	1160 (340 - 2300)	1240 (490 - 3800)
Cl mg/ℓ	98 (< 0.2 - 270)	15 (0 - 98)	57 (6.2 - 260)
Se mg/ℓ	0.31 (0.01 - 1.52)	0.02 (0.01 - 0.13)	0.59 (0.02 - 1.06)
V mg/ℓ	0.88 (0.4 - 1.3)	0.3 (0.3 - 0.3)	0.55 (0.3 - 1.3)
Radium-226 pCi/ℓ	0.47 (0.11 - 1.92)	0.71 (0.07 - 3.7)	0.22 (0.05 - 0.72)
Uranium, mg/ℓ	1.31 (0.04 - 2.6)	0.35 (0.02 - 1.0)	4.72 (0.07 - 14)
Th-230, pCi/ℓ	0.12 (0.017 - 0.52)	0.030 (0.015 - 0.053)	0.212 (0.018 - 0.65)
Th-232, pCi/ℓ	0.11 (0.0053-0.54)	0.015 (<0.01-0.036)	0.123 (0.0094-0.99)
Po-210, pCi/ℓ	0.75 (0.070 - 2.3)	0.42 (0.19 - 0.79)	0.193 (0.010 - 0.55)

[a]Mean and range of values shown.

Note.--Selenium, vanadium, and uranium values for the limestone and alluvium/
chinle aquifers are based on 4 to 5 analyses and must be regarded as tentative.

trations in mine effluents, together with low hydraulic heads, indicate short migration distances in groundwater for radionuclides and most stable trace elements in mine effluents.

Discharge of water pumped from mines to arroyos has both hydraulic and water quality impacts on shallow groundwater in the alluvial aquifer. The seepage model (Appendix H) and scattered field measurements in the Grants Mineral Belt substantiate that significant groundwater recharge is associated with mine discharge. Water quality effects on groundwater are poorly documented, however. We do not address the influence of impoundments used to remove suspended solids from mine effluents before discharge. Seepage water losses from such impoundments are believed to be small, especially when compared to infiltration losses in the arroyos and open fields receiving most of the wastes not piped to mills for process water. The impoundments are rather small and tend to become self-sealing due to settlement of fines. In at least one instance in Ambrosia Lake, the mine pond is lined to prevent seepage.

Unpublished flow and water quality data from the U.S. Geological Survey (P. Frenzel, written communication, 1979) document conditions in the Rio Puerco drainage near Churchrock and Gallup, New Mexico. Figure 3.21 shows the sampling station locations, and the chemical data are in Table 3.47. From October 1975 gaging data, seepage and evaporation reduce flow 9.39 m^3/min in a reach of 30.2 km, a loss of 0.31 m^3/min/km. Conservatively assuming 20 percent of this is by evaporation, seepage is 7.5 m^3/min or 3.94 x 10^6 m^3/yr. Gaging data for July 1977 and May 1978 similarly indicate average bed losses of 0.24 m^3/min/km. In the Ambrosia Lake district (data not shown), discharges (to San Mateo Creek and Arroyo del Puerto) from about a dozen mines total about 10.8 x 10^6 m^3/year, and the total length of perennial stream is about 15 kilometers. Assuming an average stream width of one meter and the above evaporation rate, evaporation and infiltration are 0.06 m^3/min and 7.54 m^3/min, respectively. In this case, infiltration amounts to 99 percent of total loss. Dissolved solids range from 520 to 1231 mg/ℓ (mean 743 mg/ℓ), and Ra-226 ranges from 0.2 to 23 pCi/ℓ (mean 6.6 pCi/ℓ). Selenium and molybdenum average 0.010 and 0.22 mg/ℓ, respectively.

Considering these two areas, evaporation averages about 4 percent of mine discharge versus the value of one percent calculated in Appendix H. Obviously, increased evaporation is accompanied by decreased infiltration.

Infiltration ranges from at least 90 percent to perhaps 99 percent of mine discharge, or from 1.8 to 1.98 m^3/min per mine. The foregoing field data and the more theoretical approach used in Appendix H show reasonable agreement on the relative amounts of infiltration and evaporation. We conclude then that most of the mine effluent infiltrates within relatively short distances of the mine(s) and recharges the shallow water table. The dissolved, generally nonreactive contaminants such as chloride and sulfate are expected to reach the water table, but reactive contaminants such as radium-226 and most trace metals would sorb or precipitate in the soil (substrate) in the course of infiltration.

The influence of mine discharge on groundwater quality beneath formerly ephemeral streams now receiving the discharge is currently under investigation by the New Mexico Environmental Improvement Division. Monitoring wells have been installed at several locations along the Rio Puerco (west) in the Churchrock area and San Mateo Creek in the Ambrosia Lake district. Table 3.50 summarizes partial results of samples taken in the last 12 to 18 months. In the Ambrosia Lake district, marked deterioration in water quality between the Lee Ranch and Sandoval Ranch stations on San Mateo Creek is a result of either natural causes and (or) mine drainage from a nearby deep underground uranium mine. Between Sandoval Ranch and Otero Ranch even more pronounced changes occur. In this short reach of 2.5 km, contaminated flows from uranium mines, ion-exchange plants, and seepage from an acid leach uranium mill enter Arroyo del Puerto, a tributary of San Mateo Creek. Additional study of surface water quality in the Arroyo del Puerto is recommended to further characterize the obviously interconnected surface water and groundwater systems.

In the Churchrock area, drained by the Rio Puerco, groundwater quality changes in the downstream direction are not readily apparent (Table 3.50). Although there is an acid leach mill also adjacent to the Rio Puerco tributary receiving the mine discharges, the mill is relatively new (1978 start-up) and may not yet influence stream quality. Most of the discharge from one of the two mines is used as mill feed water, thereby causing decreased discharge from the mines to the stream. Nevertheless, the reach of the perennial stream is increasing, indicating infiltration of remaining mine effluent and addition of water to storage in the shallow aquifer. Storage changes have been confirmed by static groundwater level measurements in the area east of

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Table 3.50 Groundwater quality associated with the San Mateo Creek and Rio Puerco (west) drainages in the Grants Mineral Belt, New Mexico

Station	Sulfate (mg/ℓ)	Molybdenum	Selenium (μg/ℓ)	Uranium
San Mateo Creek				
Lee Ranch	125.7	<10	< 5	<10
Sandoval Ranch	225-274	103-235	4-14.7	293-400
Otero Ranch	463-989	350-516	33-59	680-860
Rio Puerco (west)				
Hwy. 566 Bridge on N. Fork Rio Puerco	101-223	<10-284	20-22	530-760
Rio Puerco at Fourth St. Bridge, Gallup	163-244	<10-215	9-26	550-625

Source: Based on unpublished 1978 data developed by the New Mexico Environmental Improvement Division (J. Lazarus, oral communication, 1979).

Gallup. A massive spill of mill tailings into the Rio Puerco occurred in July 1979 and will complicate water quality investigation, insofar as the mine and mill influences are now superimposed in terms of both solid and liquid waste loadings in the watershed. The tailings "flood," estimated to contain about 360,000 m^3 of fluid and 1000 MT of solids, was traced into Arizona.

In summary and considering the high volume of dilute mine discharges,

which are enriched in certain stable and radioactive toxic trace elements (EPA75; Hi77), we recommend that water quality effects of mine discharge be very carefully evaluated in at least a few selected areas. Available stream-flow data indicate that infiltration is the principal means of disposal, yet the water quality data base, in particular, is rather weak to assess whether adverse impacts are likely. It is expected that future discharges in the Churchrock area alone will amount to about 40 m^3/min and will contain less than 400 mg/ℓ dissolved solids, most of which is sodium and bicarbonate. Dissolved concentrations of uranium, radium, iron, selenium, and vanadium are elevated relative to drinking water limits and infiltration of uranium, selenium, and possibly other stable elements warrants study. Use of settling ponds and barium chloride treatment greatly reduces the suspended solids, uranium, and radium concentrations. The final composition and ultimate disposal of pond sediments and added chemicals is essentially undocumented and bears additional investigation. Lastly, mine dewatering creates marked regional cones of depression and reduces the flow of water to existing supply wells and the baseflow component in major drainage systems such as the San Juan River (Ly79).

3.4.4   Gases and Dusts from Mining Activities

3.4.4.1   Radon-222 in Mine Exhaust Air

Unlike surface mines, large capacity ventilating systems are required in underground uranium mines, primarily to dilute and remove Rn-222 that emanates from the ore (Section 1.3.3). Ventilation rates vary from a few hundred to a few hundred thousand cubic meters of air per minute, and measured Rn-222 concentrations in mine vent air range from 7 pCi/ℓ to 22,000 pCi/ℓ (Ja79b). The concentration of Rn-222 in mine exhaust air varies depending upon ventilation rate, mine size (volume) and age, grade of exposed ore, size of active working areas, rock characteristics (moisture content and porosity), effectiveness of bulkhead partitions, barometric pressure, ore production rates, and mining practices. The emanation of Rn-222 dissolved in water that seeps into most mines may also contribute to Rn-222 in the exhaust air.

Because of the numerous variables that affect Rn-222 concentrations in mine air, it is difficult to confidently model radon releases from underground mines. A useful model would be one that would relate radon emissions to the production of U_3O_8. Measurements relating radon emissions to ore production have been made at seven underground uranium mines in New Mexico (Ja79b). The results of these measurements varied at the different mines from 1,380 to 23,500 Ci Rn-222 per AFR*, with an average rate of 4,300 Ci Rn-222 per AFR. The higher emission rates were noted to occur at the older mines. This was believed due to larger surface areas of exposed ore and sub-ore in the older mines. That is, inactive mined-out areas increase with mine age, and the ceiling, floors, and walls of these areas still contain certain amounts of ore and sub-ore. Radon emanating from these surface areas tend to increase the Rn-222 content of exhausted mine air unless these inactive areas of the mine (rooms, stopes, drifts, etc.) are effectively sealed. Because the radon emission factor is so variable in terms of Ci per AFR, an emission rate based on cumulative U_3O_8 mined has been proposed for modeling purposes (Ja79b). It is believed that this relationship would reduce the apparent dependence of the emission rate on the mine age. However, data are not presently available to make this latter correlation.

Although the average measured Rn-222 exhaust factor of 4,300 Ci/AFR is tentative and may be improved by studies in progress (Ja79b), it is the only value currently available for modeling purposes and will, therefore, be used in the present assessment. Assuming that 1 AFR is equivalent to 245 MT** of U_3O_8 (Ja79b), 0.017 Ci of Rn-222 will be released from the mine vents per metric ton of 0.1 percent grade ore mined. This emission rate will include all underground sources, i.e., emanation from exposed ore and blasting, slushing, loading, and transporting ore bearing rock. Radon-222 emissions were estimated for the two model underground mines by multiplying their

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*AFR = Annual Fuel Requirement for a 1000 MWe LWR.
**The AFR value on which the exhaust factor was based.

respective annual ore capacities by the above emission rate. Table 3.51 lists the results.

The estimated annual radon release computed for the average underground mine is compared below with releases reported elsewhere. Agreement is reasonably good.

Source	Annual Release of Rn-222, Ci
This Study	306
Tr79	289 - 467[a]
TVA78a	1577
TVA78b	180
TVA79	215
Th79	87

[a] Adjusted for 0.1 percent ore grade.

By properly capping the exhaust vents and sealing the shaft and mine entrance, radon emission rates from inactive mines will be a negligible fraction of the radon release rate that occurs during active mining.

3.4.4.2 Aboveground Radon-222 Sources

Radon-222 will be released from the following aboveground sources.

1. Dumping ore, sub-ore, and waste rock from the ore skip into haul trucks and unloading them on their respective piles.

2. Reloading ore from the stockpile after a 41-day residence time.

3. Emanation from waste rock, sub-ore, and ore storage pile surfaces.

The annual quantities of Rn-222 released by sources 1 and 2 were estimated using the following factors and assumptions.

* Radon-222 is in secular equilibrium with U-238.
* The density of ore, sub-ore, and waste rock is 2.0 MT/m^3.
* Annual production rates of ore and sub-ore are equal and assumed to be 1.8×10^4 MT at the average mine and 2×10^5 MT at the average large mine (Sections 3.4.1.1 to 3.4.1.3).
* The production rate ratio of ore to waste rock is 9.1:1 (Section 3.4.1.1).
* All Rn-222 present is available for release, 0.00565 Ci/m^3 per

Table 3.51   Estimated annual radon-222 emissions from underground uranium mining sources

Source	Average Mine[a], Ci/yr	Average Large Mine[b], Ci/yr
Underground		
Mine vent air	306	3,400
Aboveground		
Ore loading and dumping	1.4	15.3
Sub-ore loading and dumping	0.5	5.3
Waste rock loading and dumping	0.003	0.03
Reloading ore from stockpile	1.4	15.3
Ore stockpile exhalation	6.3	53
Sub-ore pile exhalation	61	338
Waste rock pile exhalation	0.5	2.6
Total	377	3830

[a]Annual production of ore and sub-ore = 1.8×10^4 MT, waste rock = 2.0×10^3 MT.

[b]Annual production of ore and sub-ore = 2×10^5 MT, waste rock = 2.2×10^4 MT.

percent U_3O_8 (Ni79), with an emanation coefficient of 0.27
(Au78, Tanner, A.B., Department of Interior, Geological
Survey, Reston, VA., 11/79, personal communication).
The quantities of U_3O_8 present in ore, sub-ore, and waste
rock are 0.10 percent, 0.035 percent and 0.0020 percent,
respectively (Sections 3.4.1.1 to 3.4.1.3).

Substituting the above values into the following equation yields the Rn-222 releases given in Table 3.51 for the average mine and the average large mine.

$$\text{Rn-222 (Ci/yr)} = (\text{percent } U_3O_8) \left(\frac{0.00565 \text{ Ci}}{m^3 \cdot \text{percent}}\right)(0.27) \left(\frac{m^3}{2.0 \text{ MT}}\right)$$

$$\times \left(\text{Production Rate, } \frac{MT}{yr}\right) \tag{3.11}$$

These releases are maximum values since very little time will have elapsed between the underground (blasting, slushing, loading, etc.) and surface operations. A significant amount of the radon that is available for release will emanate during the underground operations and invalidate the first assumption above concerning radioactive equilibrium. Nevertheless, these estimated maximum releases are very small in comparison to the radon released from the mine exhaust vents.

The emanation of Rn-222 from waste rock, sub-ore, and ore piles is based on an exhalation rate of 0.092 $Ci/m^2 \cdot yr \cdot$ percent U_3O_8 (Ni79) and ore grades of 0.002 percent, 0.035 percent, and 0.10 percent, respectively. Surface areas of the ore piles (Table 3.37), sub-ore piles (Table 3.38), and waste rock piles (Table 3.36) were used in these calculations. Applying these parameters, the annual Rn-222 emissions from the waste rock, sub-ore, and ore piles at the average mine and average large mine were computed. Table 3.51 gives the results. Total annual Rn-222 emissions during underground mining operations is the sum of the releases from all sources considered: 377 Ci from the average mine and 3830 Ci from the average large mine. More than 80% of the Rn-222 emissions results from the mine vent air.

3.4.4.3   Dusts and Fumes

Vehicular emissions resulting from the combustion of hydrocarbon fuels in gasoline and diesel-powered equipment are considerably less at underground mines than at surface mines (Section 3.3.4.1). The principal emissions are

particulates, sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. The quantity of these combustion products released to the atmosphere depends on the number, size, and types of equipment used, all of which are directly related to ore production.

EPA has estimated the following emissions from mining 1350 MT of ore per day from an underground mine (Re76).

Pollutant	Emissions per Operating Day, Kg/d
Particulates	2.4
Sulfur Oxides	5.0
Carbon Monoxide	41.9
Nitrogen Oxides	68.1
Hydrocarbons	6.9

Assuming a 330 operating-day year (Ni79), these emissions were adjusted according to the annual ore production of the average mine (1.8×10^4 MT) and the average large mine (2×10^5 MT). Table 3.52 lists the total airborne combustion product emissions. These emissions are small compared to those at surface mines (Table 3.30). For example, these estimates indicate that the emissions of combustion products at the average surface mine are more than 100 times greater than those at the average underground mine.

At underground mines, dust is produced by both underground and surface operations. No measurements have been made of dust concentrations in mine exhaust air. Because underground mines are wet, which greatly reduces dust production, and since a large portion of the dust produced would probably deposit underground, dust emissions from underground operations are probably relatively small. Hence, dust emissions from underground operations will not be assessed.

Aboveground sources of dust include dumping ore, sub-ore, and waste rock from the skip into haul trucks; dumping these materials onto their respective piles; reloading ore from the stockpile; using dirt haul roads by vehicular traffic; and dust suspended by the wind from the waste rock, sub-ore, and ore piles. These sources will be assessed as was done previously for surface mines (Section 3.3.4.1).

Table 3.52   Estimated air pollutant emissions from heavy-duty
                  equipment at underground uranium mines

Pollutant	Emissions, Kg/yr[a]	
	Average Mine[b]	Average Large Mine[c]
Particulates	32	350
Sulfur oxides	67	740
Carbon monoxide	560	6,210
Nitrogen oxides	910	10,100
Hydrocarbons	92	1,020

[a] Based on Re76 and 330 operating days per year.

[b] Annual ore production = 1.8×10^4 MT.

[c] Annual ore production = 2×10^5 MT.

Dust emissions will vary over a wide range depending upon moisture content, amount of fines, number and types of equipment operating, and climatic conditions. Because ore is generally wet, the relative amounts of dust produced from its mining and handling are usually small. The following emission factors were selected from those suggested by the EPA for loading and dumping operations (Hu76, Ra78, Da79):

truck loading = 2.5×10^{-2} kg/MT; and

truck dumping = 2.0×10^{-2} kg/MT.

Average annual dust emissions were estimated for the aboveground mining activities by applying these emission factors to the ore, sub-ore, and waste rock production rates of the average mine and average large mine. Table 3.53 lists the results. One-half the emission factor values were applied to ore and sub-ore because they are generally wet, except when reloading ore from the stockpile. In that case, it is assumed to have dried during the 41-day residence period (Section 3.4.1.2). Also, the emission factor for truck loading was assumed valid for loading the haul trucks from the mine skip. The dust emission for truck dumping may be high since it was based on dumping of aggregate, which would have a smaller particle size distribution than the ore, sub-ore, or waste rock (Hu76).

The movement of heavy-duty trucks is a large source of dust at most uranium mines. The magnitude of this source depends upon a number of factors, including the particle size distribution and moisture content of the road bed material, vehicular speed and distance traveled, and meteorological conditions. Emission factors for heavy-duty haul trucks (1.15 kg/VKmT) and light duty vehicles (1.03 kg/VKmt) are the same as those computed for these vehicles at surface mines (Section 3.3.4.1). Dust emissions for the movement of heavy-duty haul trucks were estimated using the appropriate emission factor and assuming --

- 31.8 MT truck capacities;
- round-trip haul distances of 1.61 km to the ore and sub-ore piles and 3.22 km to the waste rock pile; and
- the annual production rates given in Sections 3.4.1.1, 3.4.1.2 and 3.4.1.3.

Table 3.53 lists the results.

Additional dust emissions will occur from light-duty vehicular traffic along access roads. Using the emission factor derived in Section 3.3.4.1 (1.03 kg/VKmt) and assuming that there are 16 km of access roads traveled 4 times a day during the 330 operating days per year, about 22 MT of dust will be produced from this source annually. Emissions that occur during haulage road maintenance is relatively small and will not be considered.

Heavy-duty, haul truck traffic at underground uranium mines produces considerably less dust than at surface mines. This is to be expected because of the vast quantities of overburden that must be transported as well as larger ore and sub-ore capacities at surface-type mines.

The dust emissions computed above for transportation assume no effective dust control program. But, haul roads are normally sprinkled routinely during dry periods, and stabilizing chemicals are applied to roadways, usually to the ore haul roads. Dust emissions along haul roads can be reduced by 50 percent from sprinkling and up to 85 percent by the application of stabilizing chemicals (EPA77b, Da79).

Table 3.53 also lists average annual dust emissions caused by wind erosion of waste rock, sub-ore, and ore piles at the model underground mines. Emission factors, computed in Appendix I, are 2.12 MT/hectare-yr for waste rock and sub-ore piles and 0.040 kg/MT for the ore stockpiles. The first emission factor was multiplied by the waste rock and sub-ore pile surface

Table 3.53   Estimated average annual dust emissions from underground mining activities

| Source[d] | Dust Emissions, MT/yr | | | | | |
| | Average Mine [a] | | | Average Large Mine [b] | | |
	Ore[c]	Sub-ore[c]	Waste Rock	Ore[c]	Sub-ore[c]	Waste Rock
Loading truck from skip at mine shaft	0.23	0.23	0.05	2.5	2.5	0.6
Truck dumping at piles	0.18	0.18	0.04	2.0	2.0	0.4
Reloading stock-piled ore[e]	0.45	NA[f]	NA	5.0	NA	NA
Wind suspended dust from piles	0.72	4.0	0.57	8.0	22	3.0
Transportation[g]	1.0	1.0	0.23	11.6	11.6	2.6

[a] Based on annual production rates of 1.8 x 10⁴ MT of ore and sub-ore, and 2.0 x 10³ MT of waste rock.

[a] Based on annual production rates of 1.8×10^4 MT of ore and sub-ore, and 2.0×10^3 MT of waste rock.

[b] Based on annual production rates of 2×10^5 MT of ore and sub-ore, and 2.2×10^4 MT of waste rock.

[c] Assumed wet.

[d] Aboveground activities.

[e] Assumed dry.

[f] NA - Not applicable.

[g] Dust emissions from heavy-duty, vehicular traffic along ore, sub-ore, and waste rock haul roads.

areas given in Tables 3.36 and 3.38, respectively, while the second factor was multiplied by the annual ore production.

Table 3.54 shows annual contaminant emissions caused by mining activities (loading and dumping) according to source location, at the mine shaft and at the piles. Contaminant emissions were computed by multiplying the total annual dust emissions at each pile (Table 3.53) by the respective contaminant concentrations in each source--waste rock (Section 3.4.1.1; Table 3.16), sub-ore (Section 3.4.1.3; Table 3.19), and ore (Section 3.4.1.2; Table 3.19). Contaminant emissions at the site of the mine shaft were computed by multiplying the annual dust emissions of ore, sub-ore, and overburden (loading truck from skip - Table 3.53) by their respective contaminant concentrations. The three products of the multiplication were then summed to give the values listed in the 4th and 8th data columns of Table 3.54. The health impact of the sources at each location will be assessed separately in Section 6.1.

Annual contaminant emissions due to wind suspension and transport of dust are listed in Table 3.55. These values were computed by multiplying the annual mass emissions (Table 3.53) by the contaminant concentrations in waste rock, sub-ore, and ore listed in Sections 3.4.1.1, 3.4.1.3, and 3.4.1.2, respectively. The uranium and uranium daughter concentrations in dusts from all sources were also multiplied by an activity ratio (dust/source) of 2.5 (Section 3.3.1.2). Although some metals may also be present as secondary deposits, it was believed that there were insufficient data to justify multiplying their concentrations by the 2.5 ratio.

The dust emissions from vehicular traffic listed in Table 3.53 (transportation) were summed with that produced by light vehicular traffic (22 MT/yr) and considered one source of emissions. Concentrations of contaminants in haul road dust have not been measured and are not known. Some spillage of ore and sub-ore along haul roads will undoubtedly raise uranium levels in roadbed dust. As an estimate, uranium and daughter concentrations in the dust were considered to be twice background, 8ppm (2.7 pCi/g), while concentrations of all other contaminants were considered to be similar to those in the waste rock (Section 3.4.1.1). Table 3.56 shows the annual emissions computed with these assumptions.

Table 3.54 Average annual emissions of radionuclides (μCi) and stable elements (kg) from mining activities at the model underground mines

Contaminant	Average Underground Mine[a]				Average Large Underground Mine[a]			
	Waste Rock Pile Site	Sub-ore Pile Site	Ore Pile Site	Mine Site	Waste Rock Pile Site	Sub-ore Pile Site	Ore Pile Site	Mine Site
Arsenic	0.0004	0.02	0.05	0.04	0.004	0.17	0.60	0.44
Barium	0.012	0.17	0.58	0.44	0.12	1.8	6.4	4.8
Cobalt	NR[b]	0.003	0.01	0.007	NR	0.03	0.11	0.08
Copper	0.0007	0.01	0.04	0.03	0.007	0.12	0.43	0.32
Chromium	<0.002	0.004	0.01	<0.01	<0.02	0.04	0.14	<0.13
Iron	0.24	2.8	9.9	7.5	2.4	3.1	110	82
Mercury	<0.0003	ND[c]	ND	<0.001	<0.003	ND	ND	0.005
Potassium	0.28	4.5	16	12	2.8	50	175	129
Magnesium	NR	0.63	2.2	1.6	NR	7.0	25	18
Manganese	0.02	0.17	0.60	0.47	0.19	1.9	6.7	5.1
Molybdenum	0.0001	0.02	0.07	0.05	0.001	0.23	0.81	0.58
Nickel	NR	0.004	0.01	0.009	NR	0.04	0.14	0.10
Lead	0.0009	0.01	0.05	0.04	0.009	0.16	0.55	0.40
Selenium	0.0001	0.02	0.07	0.05	0.001	0.22	0.77	0.55
Strontium	0.006	0.02	0.08	0.07	0.06	0.26	0.91	0.74
Vanadium	0.004	0.25	0.89	0.65	0.04	2.8	9.9	7.1
Zinc	0.0008	0.005	0.02	0.01	0.008	0.06	0.20	0.16
Uranium-238 and each daughter	0.6	45	450	222	6	495	4,990	2,410
Thorium-232 and each daughter	0.04	0.4	6.3	2.8	0.4	4	70	31

[a] Mass emissions from Table 3.53.
[b] NR – Not reported.
[c] ND – Not detected.

Table 3.55 Average annual emissions of radionuclides (µCi) and stable elements (kg) in wind suspended dust at the model underground mines

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Contaminant	Average Large Underground Mine			Average Underground Mine		
	Waste Rock Pile	Sub-Ore Pile	Ore Stockpile	Waste Rock Pile	Sub-Ore Pile	Ore Stockpile
Arsenic	0.03	1.9	0.69	0.005	0.34	0.06
Barium	0.87	20	7.4	0.17	3.7	0.66
Cobalt	NR(a)	0.35	0.13	NR	0.06	0.01
Copper	0.05	1.3	0.49	0.01	0.24	0.04
Chromium	<0.15	0.44	0.16	<0.03	0.08	0.01
Iron	18	345	126	3.4	63	11
Mercury	<0.02	ND(b)	ND	<0.005	ND	ND
Potassium	21	550	200	4.0	100	18
Magnesium	NR	77	28	NR	14	2.5
Manganese	1.5	21	7.7	0.28	3.8	0.69
Molybdenum	0.008	2.5	0.92	0.001	0.46	0.08
Nickel	NR	0.44	0.16	NR	0.08	0.01
Lead	0.07	1.7	0.62	0.01	0.31	0.06
Selenium	0.006	2.4	0.88	0.001	0.44	0.08
Strontium	0.45	2.9	1.0	0.09	0.52	0.09
Vanadium	0.30	31	11	0.06	5.6	1.0
Zinc	0.06	0.64	0.23	0.01	0.12	0.02
Uranium-238 and each daughter	45	5,450	5,700	9	990	513
Thorium-232 and each daughter	3	44	80	0.6	8	7.2

(a)NR - Not reported.
(b)ND - Not detected.

Table 3.56 Average annual emissions of radionuclides (μCi) and stable elements (kg) from vehicular dust at the model underground mines

Contaminant	Average Large Underground Mine[a]	Average Underground Mine[b]
Arsenic	0.43	0.22
Barium	14	7.0
Copper	0.86	0.44
Chromium	<2.4	<1.2
Iron	287	145
Mercury	<0.38	<0.19
Potassium	335	170
Manganese	23	12
Molybdenum	0.12	0.06
Lead	1.1	0.53
Selenium	0.10	0.05
Strontium	7.2	3.6
Vanadium	4.8	2.4
Zinc	0.96	0.48
Uranium-238 and each daughter	129	65
Thorium-232 and each daughter	48	24

[a]Mass emissions = 47.8 MT/yr.

[b]Mass emissions = 24.2 MT/yr.

3.5 In Situ Leach Mining

Because in situ leaching of uranium (see general description in Section 1.3.4) is in its infancy, a data base for performing a detailed generic environmental assessment does not presently exist. The fact that the parameters for assessing this process are so site specific and depend upon operational procedures further impedes a generic assessment. Current research projects may help to resolve many of the present uncertainties and provide the data needed to better quantify the potential source terms (La78).

In view of the expected future expansion of this uranium mining method (Section 1.3.4), a qualitative assessment that can be modified later when additional data become available was deemed necessary. This assessment was possible because of recent laboratory experiments and field measurements at pilot-scale plants (Wy77, Ka78b, NRC78b, Tw79).

Similar to other uranium mining methods, in situ leaching also produces liquid, solid, and airborne wastes. However, the quantities of these wastes and their characteristics differ considerably from those produced at surface or underground mines. Also, because the recovery, drying, and packaging of the U_3O_8 produced is often performed at the mine site, wastes from these processes should probably be included in the mine assessment.

This assessment uses the parameters of a hypothetical "typical" commercial-sized in situ solution mine. Unlike surface or underground mines, relatively few in situ facilities exist, and they are all somewhat different because of site specificity and the rapid development of new or modified techniques. The following parameters for the hypothetical mine were based upon those of the Highland, Crownpoint, and Irigaray uranium projects and those reported by Kasper et al. (1978) (Wy77, NRC78b, TVA78b).

The Hypothetical In Situ Solution Mine

(1) Size of deposit = 52.6 hectares

(2) Average thickness of ore body = 8 m (Ka78b, NRC78b)

(3) Average ore grade = 0.06 percent U_3O_8 (Ka78b, Tw79)

(4) Mineralogy = Sandstone

(5) Ore density = 2 MT/m^3

(6) Ore body depth = 153 m

(7) Mine life = 10 years (2-yr leach period in each of 5 sectors)

(8) Well pattern = 5 spot (NRC78b, TVA78b, Ka78b)

                         Injection wells = 260

                         Production wells = 200

                         Monitoring wells = 80

(9) Annual U_3O_8 production = 227 MT (Wy77, NRC78b, Ka78b)

(10) Uranium leaching efficiency = 80 percent (Ka78b)

(11) Lixiviant = Alkaline

(12) Lixiviant flow capacity = 2,000ℓ/min (Ka78b, Wy77, NRC78b)

(13) Lixiviant bleed = 50ℓ/min (2.5 percent) (Wy77, NRC78b, TVA78b)

(14) Uranium in Lixiviant = 183 mg/ℓ (TVA78b, Ka78b, NRC78b)

(15) Calcite ($CaCO_3$) removal required = 2 kg calcite per kg U_3O_8 (Wy77)

The solid, liquid, and airborne wastes generated by this facility are described below. Wastes and quantities generated, as well as operations and procedures selected, will naturally differ to varying degrees from those at some operating sites.

3.5.1   Solid Wastes

The quantity of solid wastes generated depends upon the leachate, the ore body, and operational procedures that effect the mobilization of ore constituents. Little information is available on the quantities of solids generated because of this site dependence, the newness of the process, and the apparent relatively small quantities that are produced. Examples of solid wastes that might be expected to be generated by the alkaline leach process are listed below:

(1) Materials filtered from the lixiviant line

(2) Sediments from the surge tanks

(3) Calcium carbonate from the calcium control unit

(4) Barium sulfate from the contaminant control in the elution/precipitation circuit of the recovery process

(5) Materials deposited in the evaporation ponds

(6) Drill hole residues

(7) Solids from aquifer restoration

Sources 1 and 2

No information concerning quantities of solids from these two sources could be found in the literature, but they are described as being relatively small compared to other sources (NRC78b). These wastes are transferred to evaporation ponds and retained beneath a liquid seal.

Source 3

One of the larger sources of solids is the calcium control unit (Wy77). Calcite, $CaCO_3$, which is removed prior to injection of the refortified lixiviant, coprecipitates radium and any residual uranium. It has been reported that the amount of calcite produced is less than 2.8 kg per 1 kg of U_3O_8 recovered (Wy77). Assuming this ratio to be 2.0, and if Ra-226 is in secular equilibrium with U-238 in the ore, and 2.5 percent is solubilized by the lixiviant (Wy77, NRC78b), 454 MT of calcite will be produced annually and contain a total of 1.6 Ci of Ra-226. Also, calcite has been observed to contain between 1 to 2 percent U_3O_8 by weight (Wy77). Assuming an average of 1.5 percent U_3O_8, about 1.9 Ci (6.8 MT U_3O_8) of U-238 may also be present in the calcite waste.

Radium-226 and its daughter, Rn-222, are probably the most radiologically significant radionuclides associated with uranium mine wastes, and the small amount of Ra-226 retrieved by in situ leaching is a distinct advantage. Conventionally mining the quantity of ore assumed for the hypothetical in situ mine would contribute 64 Ci of Ra-226 per year to the surface. Because of the insolubility of $RaSO_4$, acid lixiviants containing H_2SO_4 mobilize even less radium than alkaline lixiviants. It is reported that the latter mobilizes up to 4.5 times the radium as acid leach solutions (Wy77).

If practical, the calcite waste is transferred to the mill to recover the coprecipitated uranium. Otherwise, the waste is transferred to an evaporation pond and retained beneath a liquid seal to minimize atmospheric dispersion and radon emanation.

Source 4

If necessary, the sulfate concentration in the eluant circuit of the uranium recovery unit may be controlled by the precipitation of $BaSO_4$. There are no data on the contaminant levels expected in the $BaSO_4$ waste, although less than 730 MT per year are anticipated (Wy77). These wastes are impounded beneath a liquid seal of an evaporation pond.

Source 5

An assortment of precipitation compounds will be produced by evaporative concentration of impounded waste solutions. The principal products expected are alkali chlorides, carbonates, and sulfates. The quantity of solids produced by this mechanism and their rate of accumulation on the pond bottom has not been reported.

Source 6

Residues produced from drilling the numerous wells required for in situ leaching constitute another solid waste. The hypothetical in situ leaching facility defined above requires a total of 540 wells drilled to a depth of 153 m: 200 production, 260 injection, and 80 monitoring wells. A diameter of 10.2 cm will be assumed for all wells, although 5.1 cm, 12.7 cm, and 15.2 cm diameter wells have been used (Wy77). To accommodate a concrete and steel casing, a drill hole of approximately 20 cm will be required. The residue from drilling the monitoring wells will consist mostly of barren rock; however, an equivalent of an 8-m section of each injection and recovery well will contain 0.06 percent grade ore. Hence, drill hole residues will consist of 4,960 MT of barren waste rock and 230 MT of ore containing 138 kg of U_3O_8. These wastes are in relatively small quantities and should be·manageable. The waste rock and ore, if mixed and stored in a 2-m-high rectangular pile, would only cover an area of about 0.15 hectares and average 0.0027 percent U_3O_8.

Source 7

During the active mining period, all solid wastes are generally retained beneath a liquid seal in lined evaporation ponds to minimize atmospheric dispersion and radon emanation. A plan for the final disposal of solid wastes has-not been determined. Suggested procedures are to transport the wastes to a conventional uranium mill for further treatment to recover any U_3O_8 present, treat the effluent as mill wastes, construct long-term tailings ponds on the site, or ship the wastes to a licensed off-site burial ground. Solid wastes probably comprise the least significant type waste relative to health and the environment. Solid wastes generated from reclamation procedures will be discussed in Section 3.5.5.

3.5.2 Associated Wastewater

Water flushed through the leached area when restoring the well field is the largest source of wastewater (see Section 3.5.5). The principal sources of wastewater generated by the hypothetical facility during the leaching and recovery operations are as follows:

(1) Lixiviant bleed -- barren lixiviant removed from the leach circuit to produce a net inflow into the well-field area and to control contaminant concentrations

(2) Resin wash -- water to wash resin of excess NH_4Cl used to regenerate the resin. Lixiviant bleed is sometimes used for this operation, and it reduces the total quantity of wastewater produced (Ka78b)

(3) Eluant bleed -- barren eluant removed to control salt accumulation, principally NaCl and Na_2CO_3, and maintain proper volume

(4) Well cleaning -- water used to flush injection wells to prevent clogging

The sources of wastewater and the quantities produced vary at different sites, depending upon the lixiviant and recovery circuit chemistry as well as the production rates. However, estimates were made of the quantities of wastewater generated by the four principal sources for the hypothetical in situ facility, and they are listed in Table 3.57. It is assumed that waste from backwashing the sand filters is lixiviant bleed waste water and does not contribute to the total wastewater generated. The total volume of wastewater estimated to be generated is 8.43×10^4 m^3/yr. Assuming the evaporation ponds are 3.05 m deep with a 0.604 m freeboard (Wy77) and a natural evaporation rate of 142 cm/yr (TVA78b), a pond capacity of 34,770 m^3/yr which would encompass a surface area of about 1.4 hectares/yr would be required. Using evaporation data assumed for the Irigaray Uranium Project, about 75 percent of the annual wastewater inventory would evaporate, which would leave 2.11×10^4 m^3/yr and require a surface area of 0.85 hectares/yr. If necessary, the pond size can be reduced by using mechanical evaporators.

Table 3.57 Estimated quantities of wastewater produced by an
in situ leaching operation

Source	Flow Rate, (ℓ/min)	Annual Accumulation, (m^3/yr)
Lixiviant bleed (2.5%)	50	2.63×10^4
Resin wash[a]	26	1.37×10^4
Eluant bleed	17	8.9×10^3
Well cleaning[b]	--	3.54×10^4
Total		8.43×10^4

[a]This may be included in the lixiviant bleed.

[b]Assumes 260 injection wells flushed twice each month with 5680 liters of water.

Source: Data from Wy77 and Ka78b proportioned to an annual U_3O_8 production of 227 MT and a lixiviant flow of 2000 ℓ/min; aquifer restoration is excluded (Section 3.5.5).

The liquid wastes are generally brines. They contain large amounts of sodium chloride consisting of 1,500 to 5,000 mg/ℓ total dissolved solids (TDS), trace metals ranging from 0 to 10 mg/ℓ, and small quantities of radioactivity. The quantities of contaminants generated each year were estimated for the hypothetical solution mine by using the annual mass emissions estimated for the Highland Uranium Project and adjusting the flow rates to predict the concentrations (NRC78b). Table 3.58 lists these estimated concentrations and annual emissions. Because the contaminants from the lixiviant bleed were not included in the source document, the trace metals that are mobilized by the leachate do not appear in the tabulation, and Ra-226 presence is grossly underestimated (Table 1.7, Section 1.3.4). Considering possible trace metal concentrations and their toxicities, their presence in the lixiviant bleed wastewater may be significant. Assuming that 2.5 percent of the Ra-226 in the ore is extracted, the pregnant leachate will contain about 1,520 pCi/ℓ, yielding 1.6 Ci/yr. However, it is assumed that most of this radium will be removed by the calcium control unit.

There are no planned releases of liquid wastes to the environment at in situ solution mines. The contaminants dissolved in the liquid wastes will accumulate on the pond bottoms as the liquid evaporates. Barring dike failure and seepage through the lined pond bottoms, no impact should be imposed upon the environment by this source during operation.

Another method, other than evaporation, to remove wastewater from an in situ site is deep well injection. This is the dominant method of wastewater removal at operations in South Texas (Durler, D.L., U.S. Steel Corporation, Texas Uranium Operations, Corpus Cristi, TX, 9/79, written communication).

3.5.3 Airborne Emissions

Airborne emissions from an in situ solution mining operation will originate from three principal sources: the uranium recovery and processing unit, the waste storage evaporation ponds, and the radon released from the pregnant leach surge tanks. The primary radioactive species emitted is Rn-222. The nonradioactive species emitted are a function of the lixiviant and the uranium recovery processes employed. Fugitive dust emissions, primarily from vehicular traffic, will also occur on the site. However, because very little heavy equipment is used, the potential for adverse environmental impact from this source will not be significant and is not considered in this assessment.

Table 3.58 Estimated average concentrations and annual
accumulation of some contaminants in wastewater

Contaminant	Concentration, mg/ℓ	Annual Accumulation, kg
Calcium	64	5,380
Chlorine	2,070	173,880
Carbonate	31	2,600
Bicarbonate	36	3,020
Magnesium	24	2,020
Sodium	1,320	110,880
Uranium-238	1	84
Radium-226	$21^{(a)}$	$1.8^{(b)}$
Thorium-230	$6^{(a)}$	$0.5^{(b)}$

(a)Units are pCi/ℓ

(b)Units are mCi.

Note.--Mass emissions estimated for the Highland Uranium Project (NRC78b), adjusted for flow rates and U_3O_8 production of the hypothetical solution mine.

Estimated average annual airborne emissions were computed for the hypothetical facility using data supplied by the Irigaray and Highland Uranium Projects and from the report of Kasper, et al. (1978) (Wy77, NRC78b). Table 3.59 gives the results, proportioned to a production rate of 227 MT/ yr.

The major sources of emissions from the uranium recovery plant are by-products of combustion from the dryers, volatilized solution residuals, and U_3O_8 fines generated during product drying. Carbon dioxide is the major combustion product emitted, although sulfur dioxide may also be significant if oil is used to fuel the dryers. Ammonium salts, used in the precipitation of uranium and resin regeneration, will volatilize as both ammonia and ammonium chloride during yellow cake drying. Airborne particulates that include uranium and some decay products are generated during the drying and packaging processes. The emission rates of U_3O_8 and daughter products were computed on the basis of an average release rate of 363 kg of U_3O_8 per year

Table 3.59   Estimated average annual airborne emissions from the hypothetical in situ leaching facility

Source	Annual Release Rate
Recovery Plant[a]	
Uranium-238	1.0×10^{-1} Ci
Uranium-234	1.0×10^{-1} Ci
Uranium-235	4.8×10^{-3} Ci
Thorium-230	1.7×10^{-3} Ci
Radium-226	1.0×10^{-4} Ci
Lead-210	1.0×10^{-4} Ci
Polonium-210	1.0×10^{-4} Ci
Ammonia	3.2×10^{0} MT
Ammonium chloride	1.2×10^{1} MT
Carbon dioxide	6.8×10^{2} MT
Surge Tank	
Radon-222[b]	6.5×10^{2} Ci
Storage Ponds[c]	
Ammonia	1.0×10^{2} MT
Ammonium chloride	3.0×10^{2} MT
Carbon dioxide	7.5×10^{1} MT

[a] Includes the calcium control unit.

[b] Assumes all radon formed dissolves in the lixiviant and 100 percent is released on contact with the atmosphere.

[c] Based on a release rate of 14.6 MT/yr of NH_3, 10.6 MT/yr of CO_2 and 42.0 MT/yr of NH_4Cl per hectare of pond surface (Wy77), and an average pond surface area of 7.1 hectares (1.42 ha/yr x 5 yrs).

from a 227 MT/yr facility (Wy77, Ka78b). High efficiency filters and scrubbers are used, which significantly reduce the releases from the uranium recovery plant.

Emission rates from the wastewater storage ponds are determined by the composition of the waste solutions, evaporation rate, feed rate to the ponds, and the water temperature. The principal emissions from storage ponds servicing an alkaline leach process, as defined for the hypothetical facility, are ammonia, ammonium chloride, and carbon dioxide. Different atmospheric releases would result from waste ponds servicing an acid leach facility. The release of Rn-222 from the pond surfaces has not been measured. The emission rate of Rn-222 resulting from the decay of Ra-226 contained in the pond sediments will be inhibited by the liquid seal maintained over the entire surface area of the pond. Because of its low solubility in the unagitated pond water, it is reasonable to conclude that the rate of release for radon from the water surface will be small compared to that from the pregnant leach surge tanks. The liquid seal maintained over the pond area minimizes airborne particulate emissions from the storage ponds.

The principal source of airborne radioactive emissions is the release of Rn-222 from the pregnant leach surge tanks. Rn-222 is mobilized from the ore zone during solution mining and will be largely soluble in the lixiviant under the very high pressure (~15 atm) that exists at the ore zone depth (~500 ft). Upon reaching the atmosphere at the surge tank, nearly complete release of the absorbed radon will take place. Since nearly all Ra-226 remains underground in the leach zone--only 2.5 percent is assumed to be extracted--Rn-222 will continue to be generated in areas leached of uranium.

Consider a 2-year leach period in each of 5 sectors that is 80 percent efficient and yields an average of 227 MT of U_3O_8 per year. If U-238 and Ra-226 are initially in secular equilibrium and 97.5 percent of the Ra-226 remains underground, 156 Ci of Ra-226 will be continually available for Rn-222 production. This quantity of Ra-226 will yield a lixiviant concentration in the 252,800 m^3 aquifer (Section 3.5.5) of 6.18 x 10^5 pCi/ℓ , assuming a maximum emanating power of 100 percent. The latter assumption will result in a maximum Rn-222 concentration in the lixiviant. A high emanating power is probable considering the conditions that exist in the aquifer: high pressure, high permeability due to leaching, the presence of water in the rock pores, radium present on grain surfaces, and the flow rate of water through the ore zone (Ta78, Tanner, A.B., Department of Interior,

Geological Survey, Reston, Va, 11/79, personal communication). Therefore, applying these maximizing conditions with a pumping rate of 2,000 ℓ/min, 650 Ci/yr of Rn-222 will be released at the pregnant leachate surge tanks.

Apparently very few measurements of Rn-222 concentrations in pregnant leachates have been made at operating facilities. One investigator reports that measured concentrations range from 10,000 pCi/ℓ to over 500,000 pCi/ℓ and may vary with time at the same well by factors greater than ten (Waligora, S., Eberline Instrument Corp., Albuquerque, N.M., 1979, personal communication). The concentration computed above for the model facility lies above the observed range.

3.5.4 Excursion of Lixiviant

A production zone excursion refers to the event when the leach solution flows from the leach field contaminating the surrounding aquifer. Production zone excursions are usually prevented by bleeding a small fraction (2 to 7 percent) of the lixiviant before reinjection. This imposes an imbalance in the injection-recovery volumes and causes groundwater to flow into the leach field from the surrounding stratum.

Production zone excursions are detected by wells placed 60 m to 300 m from the well field. These wells are routinely monitored, generally bi-weekly, to detect concentration increases of one or more constituents of the lixiviant. Lixiviant constituents monitored may be chloride, ammonia, bi-carbonate, sulfate, calcium, or uranium. In addition, conductivity and pH measurements are usually included. When one or more of the indicators ex-ceeds a maximum limit specified in the operator's permit, the observation is verified by resampling. If positive, sampling frequency is increased, appro-priate government agencies are notified and corrective actions are begun.

An excursion from the production zone may be terminated by one of the following suggested methods (Wy77):

(1) Overpumping - increasing the flow rate of the recovery wells to increase the inward flow of native groundwater

(2) Reordering - applying different pumping rates of the recovery wells to different areas of the well field, providing a greater inflow of native groundwater at specific points

(a variation of overpumping)

(3) Reducing Injection - another method of increasing the ratio
     of recovery flow to injection flow providing the same effect
     as overpumping

(4) Ceasing to Pump - stopping both recovery and injection flows
     (migration is then due entirely to natural groundwater flow,
     which is many orders of magnitude less than with wells pumping)

(5) Begin Restoration - initiated when all other efforts have
     failed to stop the migration of lixiviant from the leach
     field (Section 3.5.5)

Excursions are likely to occur during the operation of an in situ leach mine. Adverse consequences of an excursion will be determined by its extent, the rate of outward flow, contamination levels, aquifer hydrology, and the effectiveness of corrective measures applied.

3.5.5 Restoration and Reclamation

Restoration is the process by which the in situ leach site is returned to an environmentally acceptable state after mining is complete. Surface restoration consists of removing all structures, pipelines, and so on and sealing the evaporation ponds. Subsurface restoration, the primary area of concern, is done by discontinuing lixiviant injection and continuing pumping to sweep fresh groundwater from the surrounding area through the leached ore zone. It is anticipated that this process will flush out the remaining lixiviant and chemical compounds or elements that have adsorbed or reacted with the mineral content of the aquifer. The water recovered can be purified by chemical precipitation, ion exchange, reverse osmosis, or other processes, and then recycled. This reduces considerably the quantity of water that must be managed. Between 75 and 80 percent of the water can be reinjected while the remainder containing the contaminants is transferred to an evaporation pond (Wy77, NRC78b). During the initial restoration process, it is generally cost effective to recover the uranium from the process wastewater.

Aquifer restoration continues until the groundwater quality in the mining zone meets a criterion established on a basis of the premining water quality. In many cases, the premining groundwater quality criterion is difficult to establish because water quality can vary considerably over the ore zone region and may contain high natural levels of contaminants. Samples of water from wells monitored prior to mining in Texas contained concentrations

of Rn-222 approaching 20,000 pCi/ℓ (Tanner, A.B., Department of Interior, Geological Survey, Reston, Va, 11/79, personal communication), and it is probably unrealistic to attempt to restore an aquifer to a better quality than existed naturally before mining. Wells and flow rates used in this process must be carefully selected and controlled to provide efficient groundwater sweeps and to insure that all affected areas of the leach zone are restored.

The affected aquifer volume that is to be restored may be estimated by the following equation:

$$\text{affected volume} = \text{area of well field} \times \text{aquifer thickness} \times \frac{\text{(porosity)}}{100 \text{ percent}} . \quad (3.12)$$

Assuming a porosity for sandstone of 30 percent (NRC78b), the affected volume of the hypothetical in situ solution mine defined in Section 3.5 would be:

$$\text{affected volume} = 52.6 \text{ hectares} \times 8 \text{ m} \times 30 \text{ percent}/100 \text{ percent} = 1.26 \times 10^6 \text{ m}^3.$$

Because of mixing leach solution with the incoming sweep water and the gradual desorption of some contaminants from clays present in the ore body, more water is required to adequately flush the contaminants than one pore volume. It has been estimated that five to seven pore volumes of water would be required for adequate restoration (Wy77, NRC78b). Using the seven pore volume value and assuming that 80 percent of the sweep water is reinjected after purification, a total of 1.76×10^6 m^3 of wastewater having high TDS would be transferred to the evaporation ponds during the restoration phase. If the aquifer is swept at a flow rate of 2,000 ℓ/min, restoration would take 8 years (1.6 yr per sector), and wastewater will accumulate at about 2.22×10^5 m^3/yr during this period. With careful control, restoration can be concurrent with leaching in different areas of the well field.

Table 3.60 lists estimated average concentrations of contaminants in the restoration wastewater (NRC78b) and annual accumulation rates of the contaminants based on a flow rate of 2,000 ℓ/min. In the last column are estimates of the total mass of substances produced by restoration that would become sediments in the evaporation ponds. Data were not provided for calcium, magnesium, chloride, and ammonium ions, even though the latter two are major constituents expected from an alkaline leach process (Wy77). These concentrations reflect average values, but concentrations in the wastewater during the initial phase of the restoration process will be much higher. For

Table 3.60 Estimated average concentrations and annual and total accumulations of some contaminants in restoration wastewater

Contaminant	Concentration mg/ℓ	Annual Accumulation, Kg[a]	Total Accumulation, MT[b]
Arsenic	0.2	210	1.7
Calcium	NA[c]	NA	NA
Chloride	NA	NA	NA
Carbonate	450	473,000	3,780
Bicarbonate	550	578,000	4,620
Magnesium	NA	NA	NA
Sodium	550	578,000	4,620
Ammonium	NA	NA	NA
Selenium	0.10	100	0.8
Sulfate	150	157,000	1,250
Uranium-238	< 1[d]	< 900	< 7.2
Thorium-230	100[e]	0.10[f]	0.8[f]
Radium-226	75[e]	0.08[f]	0.6[f]
Radon-222	618,000[e]	650[f]	5,200[f]

[a] Produced only during the estimated 8-yr restoration period.

[b] Total accumulation during the estimated 8-yr restoration period.

[c] NA - Data not available.

[d] Concentration after uranium extraction.

[e] Units are pCi/ℓ.

[f] Units are Ci/yr or total curies.

Source: Concentrations based on those estimated for the Highland Uranium Project (NRC78b), adjusted for a flow rate of 2,000 ℓ/min.

Table 3.61   A comparison of contaminant concentrations in pre-mining
groundwater and pre-restoration mine water (Wy77)

Contaminant	Pre-mining Water, mg/ℓ	Pre-restoration Water, mg/ℓ
Arsenic	< 0.0025	0.021
Barium	0.12	0.069
Boron	0.16	0.283
Cadmium	< 0.005	0.014
Chromium	0.0135	0.002
Copper	0.019	0.220
Manganese	0.12	0.97
Mercury	0.0028	< 0.0002
Nickel	0.018	0.218
Selenium	0.013	1.75
Silver	< 0.005	0.015
Zinc	0.003	0.22
Lead	0.0035	0.110
Chloride	10.75	524
Ammonia	< 1.0	235
Bicarbonate	139	805
Uranium (U_3O_8)	0.098	24.4
Radium-226	27[a]	371[a]
Total dissolved solids	793	1324

[a]Units are pCi/ℓ.

example, Table 3.61 compares concentrations of substances in the groundwater before mining with those after mining but before restoration. These data are from tests conducted for the Irigaray Project (Wy77) and indicate those substances whose groundwater concentrations may be elevated by in situ leaching.

Radon emission during the restoration process has not been considered (Wy77, NRC78b, Ka78b). Because essentially all Ra-226 remains in the ore zone (about 97.5 percent), it appears reasonable to expect Rn-222 emissions to continue during restoration. A leached-out sector of the model mine will contain 156 Ci of Ra-226 in an aquifer volume of 2.53×10^5 m^3 (1.26×10^6 $m^3 \div 5$). Although no measurements have been made, it would appear that the restoration wastewater will contain about the same Rn-222 concentration as the pregnant leachate during leaching, 6.18×10^5 pCi/ℓ (Section 3.5.3). Assuming a pumping rate of 2,000 ℓ/min, a maximum of 650 Ci of Rn-222 will be released during each year of restoration, resulting in a maximum total release of 5,200 Ci during the estimated 8-yr restoration.

Restoration is presently in the experimental stages. No commercial-sized facility has reached that phase of operation. Although restoration by flushing appears feasible, there have been problems when alkaline lixiviants were used, particularly those containing ammonium ions. Ammonium is the preferred cation because sodium causes the clays to swell and plug the formation, and calcium forms an insoluble sulfate that also decreases the permeability of the formation. However, ammonium ions adsorb tightly on to clays by replacing the calcium and magnesium atoms in the clays. Montmorillonite, prevalent in the Texas mining areas, has extensive surface areas that result in very large ion-exchange capacities. Once adsorbed, the ammonium ions desorb at a very slow rate and prolong the restoration. It has been reported that after sweeping a leached ore zone with 10 ore zone volumes of water, the ammonium concentration of the water was reduced to 15 to 25 mg/ℓ (Ka78b). This concentration of ammonium may not be significant, although, under aerobic conditions, ammonium ions can be oxidized to the more toxic nitrate. In a deep aquifer, this oxidation process is not likely to occur, and, because of the very low leachability of ammonium ions from clays, any ammonium retained after restoration will move to surrounding aquifers at a very slow rate.

Several ongoing research studies are trying to solve the ammonium problem (Ka78b). Potassium is being tested as a cation replacement for ammonium

in hopes that its adsorption and swelling characteristics will be favorable. Sweep solutions enriched in calcium and magnesium are being tested to determine if they will facilitate the flushing of the ammonium ion by replacing it on the clays by ion-exchange.

Restoration of the aquifer after mining stops is in the research stage. The adequacy of the restoration process and the procedures required will depend on a number of factors: the lixiviant used, concentration of specific ions in the lixiviant, the physical character of the stratigraphic unit, and the geochemical nature of the ore deposit. Undoubtedly, research will improve the process in the next few years. If the criteria of the restoration process are met, it is unlikely that there will be any adverse environmental impact from a properly restored aquifer.

Generally, the goal of reclaiming the site surface is to return the area to a state similar to that which existed naturally before mining. This often means one suitable for livestock grazing and wildlife habitat. The following site reclamation actions have been proposed (Wy77):

(1) Remove all structures and exposed pipes and plug all wells with concrete.

(2) After all impounded liquids have completely evaporated, cover the remains with overburden to a depth [2 m has been suggested at the Irigaray site (Wy77)] that will support plant growth and suppress Rn-222 emissions or transport and deposit the remains in a mill tailings impoundment.

(3) Before backfilling, dispose of the solids containing sufficient radioactivity to warrant removal by one of the methods suggested in Section 3.5.1.

(4) Grade surfaces of the backfilled ponds and all other barren areas to create a suitable topography and then revegetate them.

(5) Irrigate and fertilize sites to develop adequate plant cover.

(6) Maintain fences to prevent grazing by livestock until stable vegetative cover becomes established.

(7) Monitor reclaimed sites for radiation, verification of vegetative cover, and the absence of adverse erosion.

(8) Sample monitoring wells one year after restoration to verify aquifer restoration.

3.6 Other Sources

3.6.1 Mineral Exploration

During early exploration, uranium was identified by its mineral color, i.e., pitchblende from the Central City District in Colorado and carnotite in the Uravan Mineral Belt in Utah and Colorado. It was usually mined in conjunction with other metals and minerals. Later, when portable radiation survey meters became available, a substantial portion of the uranium findings (generally outcrops) were made by non-geologic prospectors (UGS54). Current uranium exploration uses extensive geological studies to locate formations with a strong potential for uranium ore content. These formations are then explored and field surveyed to verify the presence of ore. Much of the current exploratory activity is directed at expanding known deposits and mining areas.

As the surface and near-surface uranium deposits are found, mined, and depleted, exploration for reserves must be conducted at greater depths. The deeper uranium deposits, however, offer few radiometric clues on the surface regarding their location. In these cases, geologic studies and field work postulate the existence of promising geological formations. Actual exploration must be done by drilling. Drilling is also used to extend and explore known uranium producing areas.

There are two categories of drilling: exploratory and developmental. Exploratory drilling is used to sample a promising formation to determine if uranium ore is present. The drilling is generally done on a grid with the drill holes spaced 60 m to 1.6 km or more apart. Development drilling, to define the size and uranium content of the ore body, occurs when ore is struck in an exploratory hole. The development hole spacing ranges from 8 m to 100 m, depending on the characteristics and depth of the ore body. Usually, the same drilling equipment is used for both the exploratory and development drilling.

Ordinarily, there are three vehicles in a drilling unit. One vehicle carries and operates the drill rig, the second carries the drill rods, and the third carries water. Although the drill rig is a well-engineered, compact design, its physical size is increasing to meet the demands of deeper drilling (Personal communication with G. C. Ritter, 1979, Bendix Field Engineering Corp., Grand Junction, CO).

Early drilling (1948-1956) was predominantly done with percussion drills. These drills could drill to depths of about 76 m using 2.8 cm diameter drill steel. The drill bit was cooled and cuttings were removed from the drill hole by forcing air down the center of the drill stem. The cuttings (chips, sands, and dusts) were carried up and out of the drill hole by the air stream with velocities of 914-1520 m per minute (Ni76). The chips and coarse sands collected near the bore hole while the fine sands drifted and deposited around the drill site. Dusts, however, were free to drift with the winds.

Rotary drilling, used for boring deep holes, generally has replaced percussion drilling. Drill stems of 7.3 cm diameter are used to bore holes to depths of about 1300 m. Stems with diameters of 11.4 cm and larger are used for drilling holes in excess of 1300 m. The rotary drill bits are cooled generally in the same manner as percussion drills. When groundwater is encountered, water is used as a drilling medium and for removing cuttings. The cuttings are removed from the drill hole in the form of a slurry or drilling mud. They are usually stored in basins, either fabricated or dug in the ground. If unavailable, water is hauled to the drill site by truck. The drilling muds and water are stored in portable tanks or an earth impoundment for recirculation. After the drilling is completed, very often the cuttings are scattered and the drilling mud left at the site. This practice has been discouraged over the past 10 years in the Uravan area (Personal communication with G.C. Ritter, 1979, Bendix Field Engineering Corp., Grand Junction, CO). In some cases, the cuttings are disposed of in a trench and covered up with earth. Drilling muds are also sometimes covered. In either case, containment of the drilling wastes does not appear to be a prevalent practice.

Development drilling is conducted if ore is struck in an exploratory hole. The offset distance (i.e, the distance between development drill holes) is dependent on the previous history of the ore body sizes in the area. Offsetting may occur as soon as ore is struck, or it may be delayed until the exploratory drilling is completed.

The ore body may be evaluated by bore hole logging or by examining and analyzing cores. Core drilling, if used, usually begins at the top of the ore horizon. Ore (cores and cuttings) removed from the bore hole are sometimes removed from the drill site. In cases where the ore is not removed from the drill site, it remains with the dry cuttings or in the drilling muds. The drill hole collar is sometimes plugged with 0.9 - 1.5 m of concrete after the bore hole has been evaluated. In some states, the drill hole must be plugged to seal off aquifers in order to minimize groundwater contamination.

3.6.1.1 Environmental Considerations

By 1977, the uranium industry had completed 101 x 10^6 meters of surface drilling, with an all-time yearly high of 12 x 10^6 meters (DOE79). From 1958-1977, about 821,900 surface holes were drilled, resulting in 87.8 x 10^6 meters of bore holes. No statistics are available on the number of holes drilled from 1948-1958, but the annual and cumulative meters drilled for that period is known (DOE79). In order to estimate the number of drill rig placements for that period, the total annual meters of drilling was divided by the annual average bore hole depth. The average depth per bore hole was estimated by plotting the average annual bore hole depths for 1958-1977 then using that data to estimate the annual bore hole depths for 1948-1957 by linear regression analysis (Fig. 3.22).

The data points in Fig. 3.22 appear to fall into two groups: 1958-1966 and 1966-1977. The average drilling depth of the 1966-1977 group of data points probably reflects the deep drilling in the Grants, New Mexico area that became significant in 1969. Using this information, the 1948-1958 average drilling depths were estimated from regression analysis using the 1958-1966 data points only. Table 3.62 is a summary of the DOE drilling data and the number of estimated bore holes by type and year.
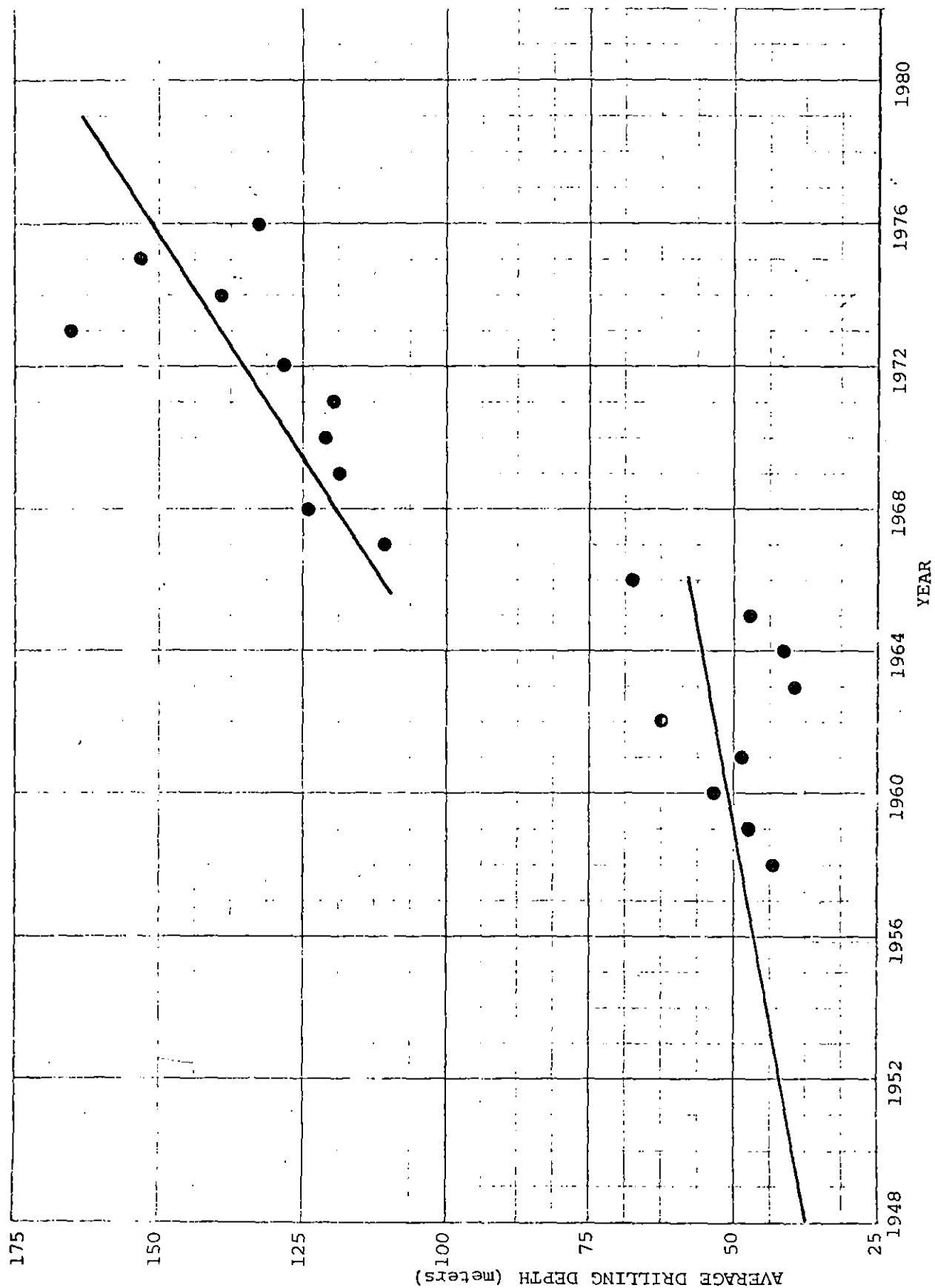
Figure 3.22 Average depth of exploratory drilling in the U.S. uranium industry from 1948 to present.

Table 3.62 Estimates of exploratory and development drill holes (1948-1979)

Year	Surface Drilling (10^6 Meters) Exploration	Development	Average Hole Depth(Meters)	Number of Holes Exploration	Development
1948	0.052	0.012	38.1[a]	1,360[b]	320[b]
1949	0.110	0.016	38.1[a]	2,880[b]	424[b]
1950	0.174	0.063	39.6[a]	4,380[b]	1,600[b]
1951	0.329	0.106	41.1[a]	8,000[b]	2,580[b]
1952	0.415	0.091	41.1[a]	10,100[b]	2,220[b]
1953	1.11	0.112	42.7[a]	26,100[b]	2,620[b]
1954	1.24	0.169	42.7[a]	29,000[b]	3,950[b]
1955	1.61	0.232	44.2[a]	36,300[b]	5,260[b]
1956	2.22	0.457	45.7[a]	48,600[b]	10,000[b]
1957	2.24	0.564	45.7[a]	49,000[b]	12,300[b]
1958	1.15	1.06	45.7	25,300	22,900
1959	0.722	1.00	48.2	16,300	19,600
1960	0.427	1.28	53.9	7,340	24,400
1961	0.402	0.972	50.0	8,260	19,300
1962	0.451	0.741	61.9	6,440	12,900
1963	0.268	0.604	39.6	8,470	13,500
1964	0.294	0.381	42.4	5,970	9,910
1965	0.354	0.289	47.5	6,230	7,330
1966	0.549	0.731	67.7	5,750	13,200
1967	1.67	1.62	110	12,800	16,900
1968	4.97	2.30	125	38,500	19,500
1969	6.25	2.86	120	47,900	28,000
1970	5.49	1.69	122	44,000	14,900
1971	3.47	1.23	121	28,400	10,400
1972	3.60	1.10	128	26,900	9,710
1973	3.29	1.70	146	22,600	11,700
1974	4.88	1.83	168	27,400	12,300
1975	5.03	2.74	139	34,300	21,600
1976	5.94	4.48	154	40,400	27,200
1977	7.89	4.45	132	62,600	30,900
1978	10.8	5.24	155[a]	69,200[b]	33,700[b]
1979	9.94	5.18	158[a]	62,700[b]	32,700[b]
TOTAL	286	149		823,000	454,000

[a] Indicates estimated average depth from Fig. 3.22.

[b] Indicates number of drill holes estimated by dividing the annual exploration and surface drilling depths by the average hole depth.

Cuttings produced by drilling can degrade the drill site area and the local air quality. For convenience of evaluation, the cuttings are divided into two general categories--dusts and wastes. The dusts are drilling fines that become airborne, and wastes are drilling chips and sands deposited around the borehole. The maximum dust production occurs when compressed air is used solely for cleaning the boreholes. Generally the drilling industry uses foaming agents injected into the compressed air stream to help remove drill cuttings. The foam traps and contains the fine particulates and substantially reduces the airborne dust. In practice, the drillers minimize airborne dust, because it causes excessive wear on engines and compressors. Dust production also indicates improper drilling energy being used to grind up cuttings in the borehole rather than bore. Occasionally some water may also be injected into the air stream to remove cuttings and to keep the drill hole from collapsing when loose materials are encountered.

There are some estimates of airborne dust production and general assumptions concerning drilling practices (Private communication with Mr. T. Price, Bendix Corp., Grand Junction, CO and E. Borgerding, Borgerding Drilling Co. Inc., Montrose, CO). They are as follows:

(1) The ratio by weight of the chips, sands, and dusts produced by drilling is approximately 60:37:3, respectively (i.e., 3 Kg of every 100 Kg of cuttings removed from a borehole is available as airborne dust).

(2) Fifty percent of all drill holes are wet (mud) drilled and 50 percent are air drilled; ninety-five percent of the latter are drilled using mist or foam (i.e., 2.5 percent are dry-drilled).

(3) The first 6.6 m of all drill holes are drilled dry (i.e., no mist or foam is used).

We estimated dust production from contemporary drilling by averaging drilling data from Table 3.62 for the years 1975 through 1979. The average depth of the holes for this period is 148 m. The annual average numbers of exploration and development holes are 53,800 and 29,200, respectively. Airborne dust production from those holes that are drilled with mud (wet), foam, or mists (97.5 percent of both the exploratory and development holes) will originate only from the first 6.6 m depth. The weight of dust generated per hole will be as follows:

Airborne dust (kg) = Volume of borehole (m^3) x density (kg/m^3) x airborne dust fraction (.03) per drill hole

$$= (\pi r^2 h) \, (2000\tfrac{kg}{m^3})(0.03)$$

where h = 6.6 m

r = 0.0865 m (assumed average rad-
ius of 2 bit sizes r = 7.3 cm
and 10 cm) (Pe79)

$$= (3.14)(7.48 \times 10^{-3}) \, m^2 \times 6.6 \, m \times 2000 \, \tfrac{kg}{m^3} \times 0.03$$

$$= 9.3 \, kg$$

The average weight of airborne dust (kg) produced from all contemporary annual drilling (first 6.6 m) is

$$83{,}000 \text{ drill holes} \times \frac{9.3 \text{ kg}}{\text{drill hole}} = 7.7 \times 10^5 \text{ kg.}$$

The annual total weight (kg) of airborne dust produced from 2.5 percent of the annual number of drill holes bored (dry) where no mud, mists, or foams are used

$$= 83{,}000 \text{ drill holes} \times \frac{148 \text{ m}}{\text{drill hole}} \times 0.025 \times 47 \, \frac{\text{kg cuttings}}{\text{m}} \times$$

$$0.03 \text{ kg dust/kg cuttings} = 4.3 \times 10^5 \text{ kg/yr.} \qquad (3.13)$$

The total weight of airborne dust produced annually from each dry-drilled borehole is 209 kg.

Assuming that each development hole penetrates the 3.6 m ore body, the total amount of airborne ore and sub-ore dust produced from development drilling annually is

$$29{,}200 \; \frac{\text{drill holes}}{\text{yr}} \times \frac{3.6 \text{ m (ore and sub-ore)}}{\text{drill hole}} \times \frac{47 \text{ kg cuttings}}{\text{m}} \times$$

$$0.03 \text{ kg dust/kg cutting} \times 0.025 = 3.7 \times 10^3 \text{ kg.} \qquad (3.14)$$

The total weight of airborne ore and sub-ore dust produced from each develop-ment drill hole (no mud, mists, or foams used) is 5.1 kg.

The estimated annual quantity of ore and sub-ore brought to the surface by contemporary drilling equals:

$$29{,}200 \; \frac{\text{drill holes}}{\text{yr}} \times \frac{3.6 \text{ m}}{\text{drill hole}} \times \frac{47 \text{ kg cuttings}}{\text{m}} \qquad (3.15)$$

$$= 4.9 \times 10^6 \, \frac{kg}{yr} \text{ or } 4.9 \times 10^3 \, \frac{MT}{yr}.$$

Most of the ore will remain at the drill site with drilling muds or with the drilling wastes around the drill holes. Since the ore most usually will be the last material removed from the boreholes, it will be deposited on the

surface of the cuttings and drilling muds. This will expose the ore to the elements and subject it to erosion.

3.6.1.2 Radon Losses from Drill Holes

When the development drill penetrates an ore body, some of the ore and sub-ore bearing formations will be exposed to air in the drill hole. Some of the radon gas produced in the ore can enter into the air in the drill hole and escape to the atmosphere. The mechanisms affecting the release rate of radon from boreholes are poorly understood. Tanner observed a wide variation in radon concentrations as a function of depth in an open borehole as compared to a closed borehole (Ta58). Tanner also noted that strong winds could significantly reduce the total radon content of an uncovered borehole. Since so little is known about radon discharges from development boreholes, radon losses in this report are assessed on a "worst case" basis using the following assumptions:

1. The drill hole is not plugged.
2. About 3.6 m of ore and sub-ore were drilled.
3. All radon released into the borehole escapes to the atmosphere.
4. The average grade of the ore and sub-ore is 0.17 percent.
5. No water accumulates in the borehole.

The surface area of the borehole passing through the ore and sub-ore body is

$$2 \pi rh = 2 \times 3.14 \times 0.0865 \text{ m} \times 3.6 \text{ m} = 2.0 \text{ m}^2. \tag{3.16}$$

The radon release rate is estimated for ore and sub-ore in the borehole using an exhalation rate of 0.092 Ci/m^2 per year per percent of U_3O_8 (Ni79). The quantity of radon (Q) per development hole escaping per unit time is

$$\frac{0.092 \text{ Ci}}{\text{m}^2 \text{ yr } \%} \times 0.17\% \times 2.0 \text{ m}^2 \times \frac{1}{3.15 \times 10^7 \text{ sec/yr}} \times 10^{12} \frac{\text{pCi}}{\text{Ci}} = 990 \text{ pCi/sec} \tag{3.17}$$

The total quantity of radon per annum escaping from all development holes drilled through 1979

$$= 4.5 \times 10^5 \text{ drill holes} \times \frac{990 \text{ pCi}}{\text{sec-drill hole}} \times 3.15 \times 10^7 \text{ sec/yr}$$

$$\times \frac{1}{10^{12} \frac{\text{pCi}}{\text{Ci}}}$$

$$= 14,000 \text{ Ci/yr} \tag{3.18}$$

The "worst case" estimate can be modified by assuming 50 percent of the holes are wet and 30 percent of the remaining holes are plugged or have collapsed. In this case, the total source term would be about 4,900 Ci/yr. Since about 31 percent of the development drill holes are at surface mines and are consumed by the pits, the annual Rn-222 release from the remaining holes will be 3,400 Ci/yr.

3.6.1.3 Groundwater

Progressively deeper holes are being drilled as the ore bodies near the surface become depleted. As the drilling depths increase, one or more aquifers may be intercepted by a drill hole, and an aquifer with poor water quality may be connected with an aquifer with good water quality. Depending on the direction of flow, the quality of water may be downgraded in a good aquifer. Most states require some plugging of the drill holes to seal the aquifer in order to maintain water quality. Adequate plugging of the drill holes requires a conscientious effort on the part of the driller and the regulatory agency. Since the movement of groundwater is relatively slow, the change in the quality of water in an aquifer will not be apparent for some time. Thus, it may take a long time to correct the quality of water in a downgraded aquifer.

3.6.1.4 Fumes

It is estimated (Pe79) that 11.2 liters of diesel fuel are needed to drill 1.0 m. In 1979, the average borehole depth was estimated to be 158 m and would require about 1770 liters of diesel fuel. This fuel would be burned at a rate of approximately 173 liters per hour. Some individual holes, however, are drilled in excess of 914 m and require 10,200 liters of diesel fuel. It is estimated that about 170 million liters of diesel fuel were consumed for all 1979 drilling.

The principal emissions from the drilling power sources are particulates: sulfur oxides, carbon monoxide, nitrogen oxides, and hydrocarbons. Because of the transient nature of the drilling, these releases are not expected to substantially lower air quality over time.

3.6.1.5 Model Drilling

About 1.3×10^6 holes have been drilled and bored for all uranium mining from 1948 through 1979 for approximately 3000 mines. This would amount to about 430 holes per mine. Thirty-six percent of the holes were for development drilling, and 64 percent were for exploratory drilling. Assuming that 50 percent of the exploratory and development holes are air drilled (see Section 3.6.1.1), the airborne dust production for an average mine may be estimated as follows:

Airborne dust from all drill holes (first 6.6 m of depth air drilled dry)

$$= 430 \text{ drill holes} \times \frac{9.3 \text{ kg}}{\text{drill hole}} = 4000 \text{ kg.} \tag{3.19}$$

Airborne dust from all dry air drilling, less the first 6.6 m, (3.20)

$$= (430 \text{ drill holes} \times \frac{209 \text{ kg dust}}{\text{drill hole}} \times 0.5 \times 0.05) - 100 \text{ kg} = 2100 \text{ kg.}$$

Airborne ore and sub-ore dust produced by dry air drilling

$$= 430 \text{ drill holes} \times 0.36 \times 0.5 \times 0.05 \times \frac{5.1 \text{ kg dust}}{\text{drill hole}} = 20 \text{ kg.} \tag{3.21}$$

Total airborne dust produced from all drilling at an average mine site

$$= 4000 \text{ kg} + 2100 \text{ kg} = 6100 \text{ Kg} = 6.1 \text{ MT.} \tag{3.22}$$

Twenty kilograms of the total dust produced will be ore and sub-ore dusts. The Rn-222 emissions from the bore holes at an average mine site would be

$$430 \text{ drill holes } (0.5)(0.36) \left(\frac{990 \quad \text{pCi}}{\text{sec-drill hole}}\right) = 7.7 \times 10^4 \frac{\text{pCi}}{\text{sec}}, \tag{3.23}$$

or 2.4 Ci/yr.

Development drill holes at a surface mine would be consumed by the pit.

Tables 3.63--3.65 show airborne particulate source terms for uranium drilling for individual drill holes and for an average uranium mine. Table 3.63 lists the airborne dust produced for each type exploratory and development borehole; Table 3.64 summarizes the quantity of airborne dust produced by all types of drilling at an average mine site; and Table 3.65 lists the pollutants emitted from a drill rig power source.

3.6.2 Precipitation Runoff from Uranium Mines

Unquestionably, overland flow or surface runoff from precipitation transports dissolved and suspended contaminants from mining areas to the offsite environment. Unfortunately, the significance of this pathway rela-

Table 3.63 Estimated source terms per borehole for contemporary surface drilling for uranium

Type of Drilling	Thickness of Ore and Sub-ore Bodies (m)	Airborne Dust Production		Airborne Ore and Sub-Ore Dust Production	
		Total(kg)	Rate(kg/min)[a]	Total(kg)	Rate(kg/min)[a]
Exploratory					
Air (dry)	NA[b]	209	0.27	NA	NA
Air (mist or foam)	NA	9.3	0.27	NA	NA
Wet (mud)	NA	9.3	0.27	NA	NA
Development					
Air (dry)	3.6	209	0.27	5.1	0.27
Air (mist or foam)	3.6	9.3	0.27	NA	NA
Wet (mud)	3.6	9.3	0.27	NA	NA

[a] Based on an air drilling rate of 11.5 m/hr.
[b] NA - not applicable.

Table 3.64   Airborne dusts produced at an average mine site
from exploratory and development drilling

Type of Drilling	Quantity of Airborne Dust (kg)
All types (first 6.6 m depth)	4,000
Air drilling (dry)	2,100
Total	6,100 kg[a]

[a] Twenty kg of the total will be ore and sub-ore dusts.

Table 3.65   Estimates of emissions from drill rig
diesel power source

Pollutant	Production Rate $(kg/10^3$ liters fuel)	Quantity[a] (kg/drill hole)	Rate[a] (kg/hr)
Carbon monoxide	12.2	20.2	1.5
Hydrocarbons	4.49	7.4	0.55
Nitrogen oxides	56.2	93	6.9
Aldehydes	0.84	1.39	0.10
Sulfur oxides	3.74	6.2	0.46
Particulates	4.01	6.6	0.49

[a] Based on a drilling rate of 11m/hr.

Source: EPA77b.

tive to uranium mines is highly site specific and poorly understood. Very few field studies of runoff from uranium mining areas have been conducted, and what field data do exist frequently relate to the combined and probably greater influences of mine water discharge and milling. Most of the NRC regulations apply to mill operations, since mining is generally exempt from the agency's charter. The EPA regulations (Environmental Radiation Protection Standards for Nuclear Power Operations; 40 CFR Part 190) applicable to the uranium fuel cycle establish dose limits for individuals to provide protection for populations living in the vicinity of uranium mills. Uranium mines are excluded, and so are liquid effluent guidelines for ore mining and dressing (40 CFR 440, Subpart E). Regulations being developed under the Resource Conservation and Recovery Act (RCRA) of 1976 apply to radioactive wastes not covered by the Atomic Energy Act of 1954, as amended. Solid and liquid waste categories will be defined in forthcoming EPA regulations developed under RCRA, but it is not anticipated that runoff from mined lands will meet the waste characteristics in the regulations. Similarly, the Federal Water Pollution Control Act Amendment of 1972, the Clean Water Act of 1977, the Safe Drinking Water Act, and State regulations in general do not address surface runoff effects of mining. Without the regulatory base, studies and field data are, not surprisingly, rather scarce. In New Mexico, the State's 208 Water Quality Management Plan calls for, among other things, improved data collection on runoff from active and inactive tailings piles and from drilling, exploration, and development activities such as access road and drill site construction (So79).

We have not estimated chemical transport by overland flow because of the limited time for the study. But, it is reasonable to expect that such transport may be quite significant in an arid and semiarid climate where much of the precipitation that does infiltrate is discharged back into the atmosphere as water vapor. This has been well demonstrated in the case of uranium mill tailings (K178). Water moving back out of the soil transports dissolved salts that are deposited on the soil surface when the carrier (water) evaporates. Subsequent precipitation further transports these salts downward into the soil and laterally to offsite areas. So-called "blooms" of salt crystals, composed mainly of sulfate and chloride compounds, characterize uranium ore bodies, mill tailings piles, and mine wastes in a number of Western States, and we must presume that such salts solubilize in runoff.

This also indicates that there may be large concentrations of contaminants available for plant uptake. Molybdenum, in particular, is one of the toxic elements on such blooms, and uranium is also highly suspect. Selenium, arsenic, and vanadium may also be present, since their anions are mobile under oxidizing conditions characteristic of the near-surface, unsaturated zone (Fu78).

Overburden has been used extensively to backfill surface mines operating since the early to mid 1970's, but this is not true at many if not most older and now inactive mines. Erosion of these piles by water and wind may present the greatest problem (Ka75). Using overburden to construct access roads and dikes distributes contaminants in the local environment and may aggravate air and water pollution. Considering that 75 percent of the overburden has a grain size exceeding 2000 μm (see Table 3.12), it is unlikely that widespread physical transport will result from overburden piles. However, using overburden for roads decreases the grain size. The association of uranium and progeny with the smaller sediment-size fractions, by a factor of 2.5, increases the potential for transport by overland flow.

Tables 3.15, 3.16, and 3.19 show stable and radioactive trace elements in ores; sub-ore, and overburden from uranium mines. Understandably, uranium, thorium, and radium are high. Arsenic, selenium, vanadium, and molybdenum are almost always closely associated with uranium. Barium, zinc, manganese, copper, iron, and potassium may also be associated in certain mineral provinces and districts. Mercury and cadmium are occasionally present (Th78). There is no consistent relationship between ore grade and trace metal content in selected New Mexico and Wyoming study areas (Wo79).

Particularly in the case of active or recently active mines, surface runoff is collected with dikes and ditches that route water to settling ponds. Water spray or chemical additives can control road dust. They are commonly used in the active mining stage, but almost never used during exploratory drilling. Grading piles to a slope of 3:1 or less also helps to reduce runoff (St78), and this practice is becoming common in Texas and Wyoming. Proper planting techniques further reduce runoff by increasing infiltration and decreasing sediment transport.

The significance of surface runoff from mining areas as a dispersal mechanism was investigated as part of this study (Wo79) (see also Section 3.2.3.2). We examined stable and radioactive trace elements in soils

affected by runoff from ore, sub-ore, and mine waste/overburden piles from one active surface mining area in Wyoming and two inactive areas (surface and underground mines) in New Mexico. Although there was evidence of offsite movement of uranium and radium at all sites, transport is limited and decreases with distance from the site. In Wyoming, pollutant releases from the mine studied do not reach nearby water courses although onsite transport of stockpiled ore as a result of precipitation runoff does occur.

A U.S. Bureau of Mines (BOM, no date) study of strip and surface mining operations and their effects in the United States involved questionnaires, literature survey, and onsite examinations of 693 selected sites, among which were uranium mines in New Mexico and Wyoming. At 60 percent of the sites, on a national basis, there were no serious problems because vegetation was reestablished and the slope of the land was gentle both before and after mining. Thirty percent of the sites had eroded to depths of 0.3 m or less, and the remainder were gullied to greater depths. There were sediments from mined lands in 56 percent of the ponds and 52 percent of the streams on or adjacent to the sample sites. Spoil bank materials ranged in pH from 3 to 5 at 47 percent of the sites and are thus not amenable to plant growth. Field observations substantiate that mined land areas, be they former forests or grasslands, did not return to the pre-mining condition. Idle land increased almost fourfold because of mining. The study concluded that natural processes need to be strongly supplemented if mined sites are to revert to former uses. Since only 6.3 percent of lands mined for uranium were reclaimed from 1930 through 1971 (Pa74), it seems reasonable to conclude that there are increased sediment loads, gullying, and poor revegetation at most older inactive mines that were poorly stabilized, if at all.

The Bureau of Mines study concluded that peak sediment loads in runoff are characteristic of areas with high intensity storms and steep slopes, particularly during and shortly after mining. Such problems are less severe in arid regions, but large quantities of sediment are discharged from mine workings, spoil heaps, and access roads. In some instances, effects of wind and water erosion on steep spoil banks in arid lands are evident many years after abandonment. In areas outside Appalachia, 86 percent of the areas investigated had sufficient runoff control, and those areas where there was a problem almost exclusively involved coal, phosphate, manganese, clay, and gold.

Incidences of radioactive contamination of local surface water have been documented for the Shirley Basin uranium mine (Utah International, Inc.) in Wyoming (Ha78). The most pronounced changes in water and stream sediment quality coincided with initial strip mining and mill processing operations. Early acid-leach solution mining also had a decided impact. Pollutant loadings from overland flow, per se, were not determined but are presumed to be minor compared to aqueous discharges from mines and mills. These findings contradict those of an earlier study (Wh76) of the same mine. Soil and vegetation collected from 1971 through 1975 at 28 stations in the vicinity of the mine were analyzed for gross alpha and gross beta (1971 to 1974) and total uranium, Ra-226 and Pb-210 (1975). The study (Wh76) concluded that--

1. concentrations of the foregoing parameters were extremely variable but reasonably consistent with previously reported information;

2. there is no evidence that radionuclide concentration in soil or vegetation collected from routine monitoring stations are changing with time;

3. concentrations of radioactivity in soil and vegetation correlate with distance from the mill area to a distance of 1.2 miles; and

4. measurable ecological effects from radiation in the environs of the Shirley Basin mine cannot be demonstrated.

The absence of statistically significant soil and vegetation contamination from the mine versus the mill is noteworthy. Overall, vegetation tends toward higher alpha and beta concentrations than soil, except at the close-in, upwind sampling areas. This selective concentration in vegetation suggests aerial deposition of contaminated dust particles on vegetation, with some additional possibility for root uptake.

Estimates of surface drilling for uranium reveal that relatively large land areas are involved. The volume of cuttings removed from borings in the period 1948 through 1979 is calculated using 286×10^6 m of exploratory drilling and 104×10^6 m of development drilling (from Table 3.62). We assumed that 30 percent of the mines are surface mines, which eliminates the borings and related debris. Thus the value of 149×10^6 (in Table 3.62) is reduced by 30 percent. Average diameter for 8.5×10^6 m of borings in the period 1948 through 1956 is 2.8 cm versus 7.3 cm for the period 1957 through 1979 (see Section 3.6.1) when 426×10^6 m of drilling took place. A sample calculation for the volume removed from borings made in the period 1975-1979 follows:

$$V = \pi r^2 h \qquad\qquad (3.24)$$
$$= (3.14)\left(\frac{7.43 \text{ cm}}{2}\right)^2 (146\text{m})$$
$$= 0.632 \text{ m}^3$$

Assuming a bulk density of 2000 Kg per m^3, each boring results in 1265 Kg of cuttings at land surface. There were 415,300 borings, resulting in 263,000 m^3 of cuttings. Assuming that the average thickness of cuttings is 0.5 m, 526,000 m^2 or 0.53 Km^2 is affected. The inclusive area affected by drilling from 1948 through 1979 is 3.6 Km^2.

Table 3.66 summarizes the surface areas affected by mine wastes, ore piles, and exploration and development activities. Maximum use was made of data developed elsewhere in this report on the number of mines, waste pile dimensions and surface areas, and the summary of exploration and development. The estimate is, at best, a first approximation and needs considerable refinement.

For example, grain size, degree of consolidation, slope, vegetative cover, and other characteristics may vary considerably between ambient soil and rock materials versus mine wastes. The latter very often occur in steep, unvegetated piles and are composed of easily-eroded, friable sandstone, boulders, and fines. It is likely, therefore, that the sediment yield on a mass per time per area basis exceeds that of the surrounding areas; thus the estimate developed below may well be on the low side.

Sediment yields from areas affected by various mining operations are roughly estimated from consideration of land areas affected and unit soil loss values for the surrounding regions. Actual values for individual tailings or waste piles may be considerably different, but refining the values given will require additional analysis beyond the scope of the present study.

Potential coal mining lands in the Northeastern Wyoming range lose soil at rates of 4.8 to 167 $m^3/Km^2/yr$ (Ke76). Upland erosion and stream channel erosion in the Gillette study area are not generally serious problems, since land dissection is presently minimal and vegetative cover is well established. The potential for increased sediment yield is large, if vegetative cover were to be reduced or eliminated and slopes steepened because of mining. Certainly, during active mining, these conditions will be at least locally present. Erosion rates of 600 to 1,100 $m^3/Km^2/yr$ from mined lands in the South Powder River Basin are expected, and they are reasonably close to

Table 3.66 Sediment yields in overland flow from uranium mining areas

Source Term	Factor	No. Installations	Cumulative Source, Km^2	Annual Sediment Loading, $m^3(a)$
Active Mines				
Underground				
Ore piles	603 m^2/mine	251 mines	0.15	143
Sub-ore piles	26,700 m^2/mine	251 mines	6.7	6385
Waste rock piles	26,700 m^2/mine	251 mines	6.7	6385
Surface				
Ore piles	4.15 x 10^3 m^2/mine	36 mines	0.15	143
Sub-ore piles	67 x 10^3 m^2/mine	36 mines	2.4	2287
Overburden piles	380 x 10^3 m^2/mine	36 mines	13.7	13056
Inactive Mines				
Underground				
Waste piles and sub-ore	4.07 x 10^3 m^2/mine	2108 mines	0.86	820
Surface				
Overburden and sub-ore	6.73 x 10^4 m^2/mine	944 mines	64	61000

Table 3.66 (Continued)

Source Term / Factor		No. Installations	Cumulative Source, Km^2	Annual Sediment Loading, $m^{3(a)}$
Exploration and Development				
Drilling				
1948-1979	435×10^6 m	1.28×10^6 borings	3.6	3431
1975-1979	1265 kg/boring	415,300	0.53	506
Access roads and				
pads	1.25 acres or	1.28×10^6	6500	6.2×10^6
	0.5 ha/boring			

(a)Assumes average sediment yield of 953 m^3/Km^2.

Note.--Data in this table are based on average mine vs. average large mine as defined in Section 3 of report.

natural, pre-mining conditions (R. Loeper, Soil Conservation Service, 1979, personal communication). At the Bear Creek mine, the reclamation design calls for maximum losses from overburden piles of 1,100 $m^3/Km^2/yr$ initially and 600 $m^3/Km^2/yr$ after the first 3 years. In general, erosion and soil loss from uranium mining in this part of Wyoming is not a significant problem, mainly because of reclamation by industry. Sediment yields in the Grants Mineral Belt range from 95 to 240 $m^3/Km^2/yr$ in the area of the large Jack-pile-Paguate surface mine to 500 to 1,400 $m^3/Km^2/yr$ near the underground mining centers around Smith Lake, Ambrosia Lake, and Churchrock (P. Boden, Soil Conservation Service, 1979, personal communication). Considering both the Wyoming and New Mexico model mine areas, this study used an overall average annual soil loss rate of 953 m^3/Km^2. This average sediment yield rate is based on studies by the Soil Conservation Service of large areas in New Mexico, Wyoming, and other Western States.

In summary, the total land area directly affected by uranium mining is about 6600 Km^2. Assuming an overall average sediment yield of 953 m^3/Km^2, annual sediment transported by overland flow is approximately 6.3×10^6 m^3. Obviously exploration and development activities affect the greatest area (6500 Km^2), but they do not necessarily have the greatest impact. Exploration and development, for example, affect large areas, but most of the area affected is a result of constructing access roads and drill pads. Whereas sediment yields from ore, sub-ore, overburden, and waste rock is estimated at 90,000 m^3 per year. Surface mining, although it supplies only about 30 percent of U.S. production, affects the second greatest area (80 Km^2). We have not attempted to characterize the quality of sediment runoff. The fate of these sediments is very poorly understood and has not been the subject of intensive investigation. Further study in the area of intensive surface mining such as in Texas and Wyoming is needed to determine changes in erosion rates resulting from mining and to quantify the contaminant flux and fate.

3.7 Inactive Mines

3.7.1 Inactive Surface Mines

For generic purposes, a model inactive open pit or surface uranium mine must be defined in order to estimate the environmental impact from this type

of mining. We have assumed that an inactive surface mine has a single hole or pit in the ground, with all of the materials (wastes) stacked into piles adjacent to the pit area. The size or volume of the pit would be approximately equal to the volume of the ore and wastes removed from it. Since only 6.3 percent of all of the land used for uranium mining has been reclaimed from 1930 through 1971 (Pa74), no credit for reclamation is given to the model mine.

Ideally, the model mine size could be established by averaging the ore and waste production for each inactive surface mine. Unfortunately, these statistics are either not thoroughly documented or they are retained as company confidential information. In lieu of specific information, the model surface mine size was established from annual ore and waste production statistics for all surface mines, divided by the number of inactive surface mines.

Table 3.67 is a summary of inactive mines, obtained from the Department of Energy mine listing. The mines are listed by type, surface and underground. Most of the inactive surface mines are in Colorado, Utah, Arizona and New Mexico. For model derivation purposes, we assumed that there are presently 1250 inactive surface uranium mines.

Table 3.68 lists mine waste and ore production information from 1932 to 1977. Uranium mine waste and ore production statistics, on an annual basis, were available from both surface and underground uranium producers from 1959 to 1976 (DOI59-76). Annual uranium ore production statistics for each uranium mining type (surface and underground) are available for 1948 to 1959 (DOE79) and for combined uranium production from 1932 to 1942 (DOI32-42). In order to estimate waste production for the years prior to 1956, the annual mine type ore production records were multiplied by waste-to-ore ratios. These ratios were estimated from published 1959 to 1976 ore and waste production statistics (DOI59-76). Very little uranium ore was mined from 1942 to 1948, since most of the uranium was obtained by reprocessing vanadium and radium tailings. (personal communication with G. Ritter, Bendix Field Engineering Corp., Grand Junction, CO, 1979). The annual waste production for surface mining from 1948 to 1959 was estimated by extrapolating known waste-to-ore ratios (1959 to 1976) through the 1948 to 1959 time period using a "best fit" regression analysis (Fig. 3.23). This method cannot be used to estimate waste-to-ore ratios because the waste production is finite and will always occur, and also surface mining for uranium essentially began in 1950.

Table 3.67   Consolidated list of inactive uranium producers by
State and type of mining

State	Surface	Underground	Percent of Total Surface Mines	Percent of Total Underground Mines
AL	0	1	0.0	< 0.1
AZ	135	189	11	9.3
CA	13	10	1.0	0.49
CO	263	902	21	44
ID	2	4	0.16	0.20
MT	9	9	0.72	0.44
NV	9	12	0.72	0.59
NJ	0	1	0.0	< 0.1
NM	34	142	2.7	7.0
ND	13	0	1.0	0.0
OK	3	0	0.24	0.0
OR	2	1	0.16	< 0.1
SD	111	30	8.9	1.5
TX	38	0	3.0	0.0
UT	378	698	30	34
WA	13	0	1.0	0.0
WY	223	32	18	1.6
Total	1246	2031		

Table 3.68 Uranium mine waste and ore production (MT x 1000)

Year	Surface Mining		Underground Mining		Surface Mining Waste/Ore	Underground Mining Waste/Ore	Total Ore Produced By Surface and/or Underground Mines
	Crude Ore	Waste	Crude Ore	Waste			
1977	5059	237800	4305	3487	47	0.81	9364
1976	4238	190700	3569	2605	45	0.73	7807
1975	3809	139700	2485	2195	37	0.88	6295
1974	3510	129700	2222	1424	37	0.64	5732
1973	3800	182300	1614	934	48	0.58	5414
1972	3447	155100	2439	593	45	0.24	5886
1971	2656	120200	2836	858	45	0.30	5492
1970	2490	76870	3304	962	31	0.29	5794
1969	1653	81000	3171	1184	49	0.37	4824
1968	1989	31360	3382	1163	16	0.34	5371
1967	1393	32510	2897	1024	23	0.35	4290
1966	905	24400	2777	863	27	0.31	1768
1965	1630	17710	3055	809	11	0.26	4685
1964	2344	26680	3227	941	11	0.29	5571
1963	3578	33120	3575	946	9.0	0.26	7153
1962	2895	44640	4892	1087	15	0.22	7787
1961	3051	42500	5017	1117	14	0.22	8068
1960	2691	73570	5104	1868	27	0.37	7795
1959	2494	46790	3796	941	19	0.25(b)	6290
1958	2139	19240	2558	690	9.0(a)	0.27(b)	4697
1957	1462	11700	1888	510	8.0	0.27	3350
1956	1131	9048	1595	414	8.0	0.26	2726
1955	339	2650	1043	271	8.0	0.26	1382
1954	241	1930	762	198	8.0	0.26	1003

Table 3.68 (continued)

Year	Surface Mining Crude Ore	Waste	Underground Mining Crude Ore	Waste	Surface Mining Waste/Ore	Underground Mining Waste/Ore	Total Ore Produced By Surface and/or Underground Mines
1953	162	1300	503	126	8.0	0.25	665
1952	59	472	341	85	8.0	0.25	400
1951	25	203	289	73	8.0	0.25	314
1950	21	167	207	50	8.0	0.24	228
1949			156	37		0.24	156
1948			34	8.3		0.24	34
1947			0			0.24	
1946			0			0.23	
1945			0			0.23	
1944			0			0.23	
1943			0			0.22	
1942			0			0.22	
1941			0.824	0.181		0.22	0.824
1940			0.7221	0.151		0.21	0.722
1939			5.68	1.19		0.21	5.68
1938			3.89	0.817		0.21	3.89
1937			1.55	0.310		0.20	1.55
1936			1.31	0.261		0.20	1.31
1935			1.03	0.207		0.20	1.03
1934			0.230	0.0461		0.20	0.230
1933			0.047	0.00896		0.19	0.047
1932			0.0553	0.0105		0.19	0.0553

(a) Waste to ore ratios from 1950 – 1958 estimated from 1959 – 1972 ratios.
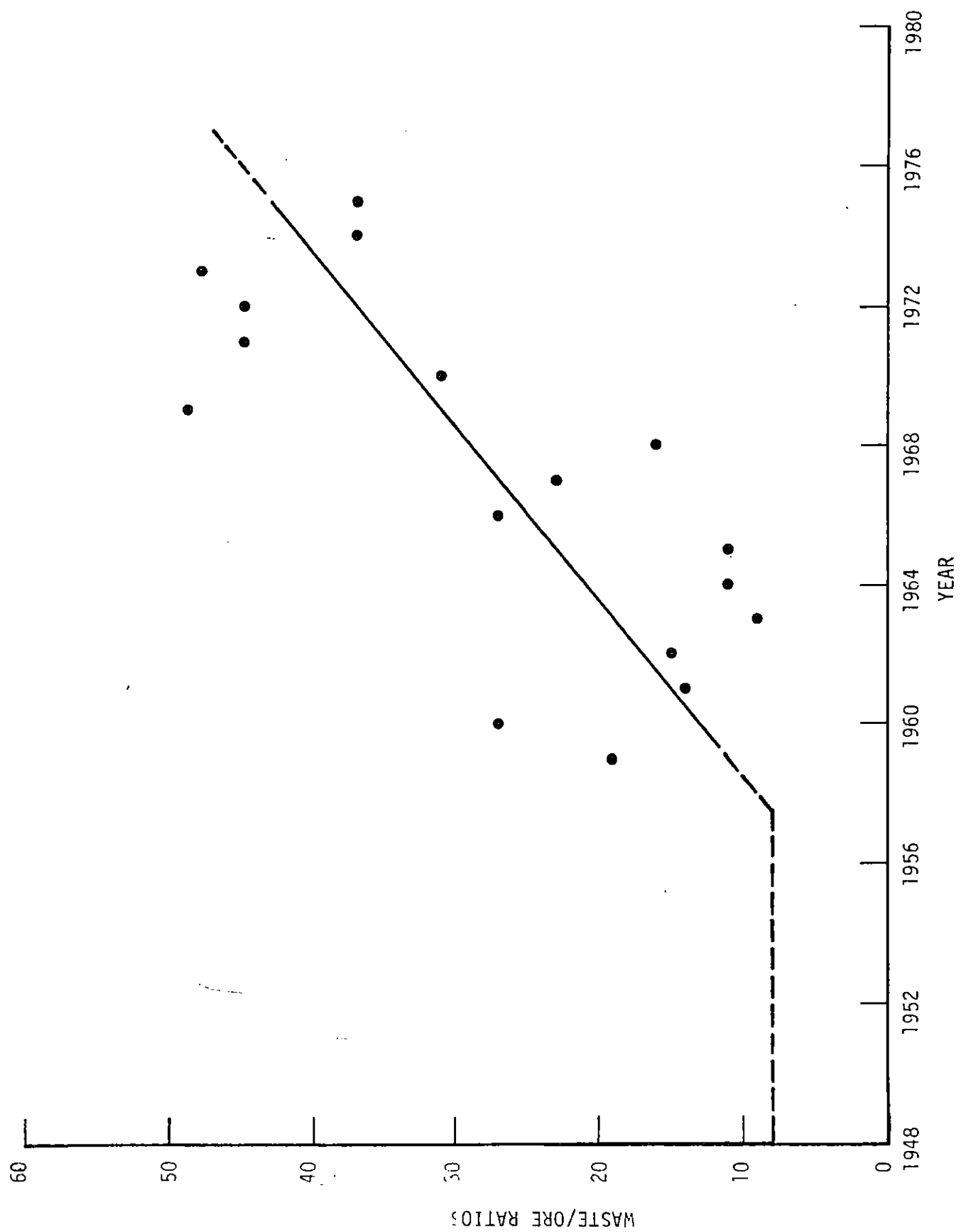(b) Waste to ore ratios from 1932 – 1958 estimated from 1959 – 1972 ratios.

Figure 3.23 Annual waste to ore ratios for surface mining of uranium (1948 to 1979).

Since early surface mines recovered ore bodies very close to the surface, the ore-to-waste ratio would be expected to be relatively small. A range of waste to ore ratios of 8:1 to 35:1 for surface mining has been estimated (C174). The lower ratio was selected to be typical for surface mining from 1948 to 1957 and was used to estimate the waste production for that period. The increase·in waste-to-ore ratios from 1959 to 1976 was probably due to several reasons. The gradual depletion of near surface ore deposits required mining deposits at increasing depths, and the development of surface mining equipment now permits economical recovery of ore at greater depths below grade. The waste-to-ore ratios for 1976 to 1977 were projected with the previous regression analysis line fit.

The estimated annual cumulative waste production from uranium surface mining for 1950 to 1978 (Table 3.69) is 1.73×10^9 MT. A crude estimate of the waste production for the model inactive surface mine can be made by dividing the total waste produced to 1978 by the number of inactive mines. But, this overestimates waste production because some of the contemporary wastes are being produced by active mines, and the waste production per mine has increased with increasing contemporary waste-to-ore ratios. To adjust the contemporary waste production for the active mines and the increasing waste-to-ore ratios, we assumed a cutoff date of 1970, based on the description of a contemporary active surface mine (Ni79). The model mine age is about 1 year as of June 1978, and has an expected life of approximately 17 years. Those mines that were active in 1970 are all assumed to have become inactive between 1970 and 1978. Their percentage of the annual waste of about 12.5 percent was assumed to decrease linearly with time from 1970-1978. For example, all of the wastes produced by surface mines in 1970 (i.e., 7.69×10^7 MT) were produced by surface mines that would be inactive by 1978. The waste production for the following years (1971-1977) was: 1.05×10^8 MT in 1971; 1.16×10^8 MT in 1972; 1.14×10^8 MT in 1973; 6.49×10^7 MT in 1974; 5.24×10^7 MT in 1975; 4.77×10^7 MT in 1976; 2.97×10^7 MT in 1977. The ore production was calculated in the same manner as for the wastes and was 3.27×10^7 MT in 1970. The ore production for the following years was: 2.32×10^6 MT in 1971; 2.58×10^6 MT in 1972; 2.38×10^6 MT in 1973; 1.76×10^6 MT in 1974; 1.43×10^6 MT in 1975; 1.06×10^6 MT in 1976, and 6.32×10^5 MT in 1977. The adjusted cumulative wastes from surface mining from 1950-1978 was 1.11×10^9 MT, and the adjusted cumulative ore production was 4.49×10^7 MT.

Table 3.69 Cumulative uranium mine waste and ore production

Year	Waste (10³MT) Surface	Waste (10³MT) Underground	Ore (10³MT) Surface	Ore (10³MT) Underground
1977	1733000	29250	59220	73100
1976	1496000	24950	54160	68840
1975	1305000	21380	49920	65210
1974	1165000	19180	46110	62760
1973	1036000	17760	42600	60500
1972	853200	16820	38800	58960
1971	698100	16240	35350	56510
1970	577800	15370	32700	53600
1969	501000	14410	30200	50330
1968	420000	13220	28550	47160
1967	388600	12060	26560	43810
1966	356200	11040	25170	40910
1965	331800	10180	24260	38090
1964	314000	9369	22640	35000
1963	287300	8425	20290	31750
1962	254200	7479	16720	28210
1961	209600	6391	13810	23310
1960	167100	5273	10770	18320
1959	93510	3406	8075	13150
1958	46720	2466	5580	9433
1957	27470	1776	3442	6839
1956	15770	1266	1979	4943
1955	6720	852	848	3356

Table 3.69 (Continued)

Year	Waste (10^3MT)		Ore (10^3MT)	
	Surface	Underground	Surface	Underground
1954	4071	580	509	2313
1953	370	171	46.3	702
1952	842	257	105	1043
1951	370	171	46.3	702
1950	167	98.9	20.9	413
1949		48.6		206
1948		11.4		49.8
1947		3.18		15.3
1946		3.18		15.3
1945		3.18		15.3
1944		3.18		15.3
1943		3.18		15.3
1942		3.18		15.3
1941		3.18		15.3
1940		3.00		14.5
1939		2.85		13.8
1938		1.67		8.12
1937		0.844		4.24
1936		0.533		2.68
1935		0.272		1.37
1934		0.0656		0.333
1933		0.0195		0.102
1932		0.0105		0.0553

Using these adjusted waste and ore values, the model inactive uranium surface mine produced 8.88×10^5 MT of waste and 3.59×10^4 MT of ore.

The volume of the remaining pit of the model surface mine would be equal to the total of the volume of wastes and ore that were removed from the mine. Assuming a density of 2.00 MT/m^3, the volume of wastes and ore removed from the mine pit would be 4.44×10^5 and 1.80×10^4 m^3, respectively. The pit was assumed to have the shape of an inverted truncated cone with a wall angle of 45° (Fig. 3.24). The ore body was assumed to be a solid right cylinder with a radius of 43.7 m and height of 3.0 m. The pit depth (ground surface to bottom of ore bed) was 36.7 m, and the ground surface area of the pit opening was calculated to be 2.03×10^4 m^2.

3.7.1.1 Waste Rock Piles

Overburden and sub-ore wastes from surface mines have been handled in several ways in the past. In one case, the sub-ore (generally the last material removed from the pit) was piled on top of the overburden. In another case the sub-ore was piled separately and blended with higher grade ore for shipment to the ore buying stations or mills. If the quantity of sub-ore was in excess of that required for blending, it was also dumped on top of the overburden (personal communication with G. Ritter, Bendix Field Engineering Corp., Grand Junction, CO, 1979). The earlier surface mining practices, therefore, generally produced waste piles with their cores containing overburden and their outer surface containing a mixture of overburden and sub-ore.

The actual method of removing and stacking overburden and sub-ore varies from mine to mine. In many cases the wastes were dumped in depressions or washes or stacked in more than one pile. For calculation purposes, we assume that wastes are stacked on a single pile in the shape of a solid truncated cone 10 m high with a 45 degree slope. It is further assumed that the sub-ore removed from the pit is placed evenly on top of the stacked overburden. The area and depth of the sub-ore placed on the waste pile is estimated by determining the areas of the base and top of the pile by iteration, computing the exposed surface area of the pile, computing the volume of the sub-ore, and calculating the depth of the sub-ore.

The areas of the base and top of the waste pile (truncated cone) were determined from the following equation:
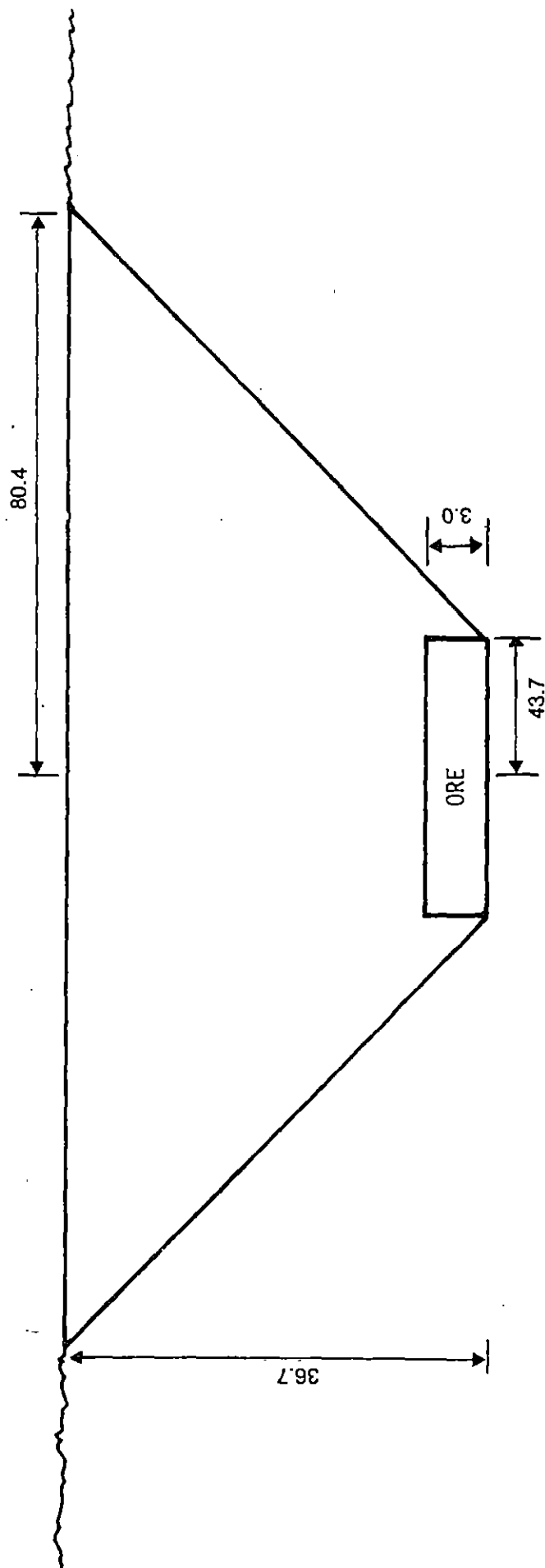
Figure 3.24 Cross section of model inactive surface mine (meters).

$$V = \frac{h}{3} (A_B + A_T + \sqrt{A_B A_T})$$ where V = volume of wastes (overburden   (3.25)
and sub-ore) (m^3)

A_B = area of the base (m^2)

A_T = area of the top (m^2)

h = perpendicular distance between the base and top (10 m)

Different values of A_B were substituted into the equation until the value of V was equal to the combined volumes of the overburden and sub-ore (i.e; 5.55 x 10^5 m^3) using a bulking factor of 25%. The area of the cone top was computed (assuming a 45 degree slope) from the diameter of the top (D_T), which is equal to the diameter of the base (D_B), minus 20 meters or $D_T = D_B - 20$. The calculated diameters, D_T and D_B, are 256 m and 276 m, respectively.

The exposed surface area of the waste pile was calculated using the following equation:

$$S = S_L + S_T$$ where S_L = lateral surface area (m^2)   (3.26)

$$S_L = \frac{L}{2} (C_B + C_T)$$

and S_T = area of the top (m^2)

$$= \pi r_T^2$$

$$S = \frac{L}{2} (C_B + C_T) + \pi r_T^2$$ where C_B = circumference of the base (m)

C_T = circumference of the top (m)

L = slant height (m)

r_T = radius of the top (m)

$$S = \frac{14.1}{2} (\pi D_T + \pi D_B) + \pi r^2$$ where D_T = diameter of the top (m)

D_B = diameter of the base (m)

$$S = \frac{14.1}{2} (3.14) (256 + 276) + 3.14 (16384)$$

$$S = 6.33 \times 10^4 \ m^2 \ \text{(exposed surface area of waste pile)}$$

The volume of sub-ore removed from the pit is assumed to be equal to the volume of ore removed from the pit. The thickness (T) of the sub-ore plate on the overburden is--

$$T = \frac{Vo}{S} = \frac{2.25 \times 10^4 m^3}{6.33 \times 10^4 m^2} = 0.36 m.$$

(3.27)

In summary, the waste pile produced at an inactive uranium surface mine is to be in the shape of a truncated cone having a surface area of 6.33×10^4 m^2. The pile is assumed to have an inner-core of overburden plated with 0.36 m of sub-ore on its exposed surface. In practice, the plate would be a mixture of overburden and sub-ore with the sub-ore concentrations increasing towards the pile surface.

Table 3.70 lists average annual emissions of contaminants due to wind erosion of the overburden pile. To compute these values, an emission factor of 0.850 MT/hectare-yr, computed in Appendix I, was multiplied by the pile surface area, 6.33 hectares, and the stable element concentrations listed in Table 3.19. Uranium and thorium concentrations were assumed to be 110 pCi/g and 2 pCi/g, respectively.

3.7.1.2 Radon-222 from the Mine Area

After the termination of active mining, Rn-222 will continue to exhale from the wall and floor of the pit. Since all of the ore has been removed, the Rn-222 will originate from the overburden and sub-ore surfaces. The surface area of the sub-ore region of the pit is estimated from the volume of ore and sub-ore (3.6×10^4 m^3) and the shape and size of the pit using the following equations:

$$V = 1/3\ h\ (A_T + A_B + \sqrt{A_T A_B})\ \text{where:}\ A_T = \pi\ r_T^2 \qquad (3.28)$$

$$S = 1/2\ L\ (C_B + C_b) + A_B \qquad A_B = \pi r_B^2 \qquad (3.29)$$

The terms in the equations are defined in the previous Section. By substituting the terms $r_B + h$ for r_T in Equation 3.28, h can be solved by iteration.

$$V = 3.6 \times 10^4\ m^3 = 1/3\ h\ [\pi\ (r_B + h)^2 + \pi\ r_B^2 +$$

$$\sqrt{\pi\ (r_B + h)^2\ (\pi\ r_B^2)}\ ]\ \text{when h = 5.3 m}$$

The exposed surface-area of the pit that contains the sub-ore is

$$S_s = 1/2\ (7.50)\ \pi(87.4 + 98.0) + 6000 = 8.18 \times 10^3\ m^2,$$

and the surface area of the overburden section of the pit is

$$S_o = 1/2\ (44.4)\ (\pi)\ (98.0 + 161.0) = 1.81 \times 10^4\ m^2.$$

Table 3.71 shows the results of radon flux measurements made at 20 of the tailings piles at inactive uranium mill sites. Also shown is the estimated average Ra-226 content of the tailings and the average Ra-226 content

Table 3.70 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive surface mine

Contaminant	Overburden Pile[a]	Contaminant	Overburden Pile[a]
Arsenic	0.46	Molybdenum	0.62
Barium	4.9	Nickel	0.11
Cadmium	ND[b]	Lead	0.42
Cobalt	0.09	Ruthenium	ND
Copper	0.33	Selenium	0.59
Chromium	0.11	Strontium	0.70
Iron	84	Vanadium	7.6
Mercury	ND	Zinc	0.16
Potassium	135	Uranium-238 and each daughter	1480
Magnesium	19	Thorium-232 and each daughter	11
Manganese	5.2		

[a] Mass Emissions = 5.38 x 10^6 g/yr.
[b] ND - Not detected.

Table 3.71 Average radon flux of inactive uranium mill tailings piles

Location	Average Radon Flux[a] (pCi/m^2-sec)	Estimated Ra-226[b] Tailings Content (pCi/g)	Average Ra-226 Background Soils[a] (pCi/g)	Reference[a]
ARIZONA				
Monument Valley	20	50	0.95	FBD-GJT-4 (1977)
Tuba City	193	924	0.95	FBD-GJT-5 (1977)
COLORADO				
Durango	197	840	1.48	FBD-GJT-9 (1977)
Grand Junction	359	784	1.52	FBD-GJT-9 (1977)
Gunnison	470	420	1.48	FBD-GJT-12 (1977)
Maybell	86	252	1.52	FBD-GJT-11 (1977)
Naturita	1446	756	1.48	FBD-GJT-8 (1977)
New Rifle	458	504	1.52	FBD-GJT-10 (1977)
Old Rifle	553	980	1.52	FBD-GJT-10 (1977)
Slick Rock	70	171	1.48	FBD-GJT-7 (1977)
IDAHO				
Lowman	125	---	1.12	FBD-GJT-17 (1977)
NEW MEXICO				
Ambrosia Lake	173	760	1.02	FBD-GJT-13 (1977)
Shiprock	340	700	1.7	Bernhardt et al. (1975)
OREGON				
Lakeview	660	420	0.81	FBD-GJT-18 (1977)
SOUTH DAKOTA				
Edgemont	143	---	1.33	FBD-211 (1978)
TEXAS				
Falls City	65	448	0.93	FBD-GJT-16 (1977)
Ray Point	430	518	0.93	FBD-GJT-20 (1977)
UTAH				
Green River	77	140	1.43	FBD-GJT-14 (1977)
Mexican Hat	290	784	0.83	FBD-GJT-3 (1977)
Salt Lake City	1200	896	1.4	Bernhardt et al. (1975)
WYOMING				
Spook Site	1770	356	0.99	FBD-GJT-15 (1977)
Average All Sites	466	563	1.26	

[a] FBD77.
[b] Sw76.

measured in representative background soils for each site. The average radon exhalation rate per average Ra-226 content of tailings material from these data is 0.83 pCi of Rn/m^2-sec per pCi of Ra/g.

Data analysis by Schiager (Sc74) indicates a radon exhalation rate of 1.6 pCi of Rn/m^2-sec per pCi of Ra/g. This value has often been used in the environmental impact statements to assess the radon flux from tailings materials.

Table 3.72 summarizes data obtained during radiological surveys of inactive uranium mine sites in New Mexico and Wyoming during the spring of 1979 (Wo79). Radon exhalation rates were measured with charcoal cannisters and the radium-226 concentrations were determined for composite surface samples taken from overburden, sub-ore, and waste rock piles. The average radon-222 exhalation rate per average radium-226 content of the overburden, sub-ore, and waste rock piles was 0.27, 0.11, and 0.12 pCi of Rn/m^2-sec per pCi of Ra-226/g, respectively.

Measurements of the background flux and Ra-226 content of typical background soils were reported for the Edgemont, South Dakota site (FBD78). These data indicate a value of 1.05 pCi of Rn/m^2-sec per pCi of Ra/g. Table 3.73 summarizes background radon flux estimates for several regions of the United States. Considering the average U.S. background flux to be 0.82 pCi of Rn/m^2-sec (Tr79) and the average U.S. background soil Ra-226 content to be 1.26 pCi of Ra/g (Oa72), the average U.S. background radon exhalation rate is estimated to be 0.65 pCi of Rn/m^2-sec per pCi of Ra/g. The average background radon exhalation rate for New Mexico and Wyoming (Table 3.72) was 0.33 pCi of Rn/m^2-sec per pCi of Ra/g. Therefore, the grand average U.S. background radon exhalation rate has been estimated to be 0.68 pCi of Rn/m^2-sec per pCi of Ra/g, and the grand average U.S. background soil Ra-226 content has been estimated to be 1.6 pCi/g.

We estimated the total radon released from the model abandoned surface mine area from the following parameters:

1. Radon exhalation from the sub-ore surface area of the pit--
   - the exposed sub-ore surface area (S_s) = 8.18 x 10^3 m^2;
   - the average radium-226 content of the sub-ore = 110 pCi/g; and
   - the radon flux rate for sub-ore = 12 pCi of Rn/m^2-sec.

Table 3.72. Average radon flux measured at inactive uranium mine sites

Location	Area	Average Radon Flux (pCi/m^2-sec)	Number of Flux Measurements	Average Radium-226 Content of Surface Sample (pCi/g)
Underground Mines				
San Mateo Mine, New Mexico	Waste pile	18	11	117
	Heap leach pond	38	3	81
	Background	0.29	1	0.77
Barbara J # 1 Mine, New Mexico	Waste pile	7.9	6	110
	Background	0.41	1	3
Surface Mines				
Poison Canyon 1, New Mexico	Sub-ore	7.0	1	43
	Overburden piles	6.7	5	62
	Background	0.33	1	2.1
Poison Canyon 2, New Mexico	Sub-ore	5.3	3	---
	Overburden pile	9.8	6	---
Poison Canyon 3, New Mexico	Sub-ore	11	2	---
Morton Ranch (Pit 1601), Wyoming	Sub-ore	24	12	170
	Overburden	9.7	4	23
	Background	2.3	2	3
Grand Averages	Sub-ore	12		110
	Overburden	8.7		32
	Waste Rock	13		110
	Background	0.83		2.2

Source: Wo79.

Table 3.73 Background radon flux estimates

Location	Radon Flux pCi/m^2-sec
Background Soils of the U.S.	
Champaign County, Illinois	1.4
Argonne, Illinois	0.56
Lincoln, Massachusetts	1.3
Socorro, New Mexico	0.90
Socorro, New Mexico	1.0
Socorro, New Mexico	0.64
Yucca Flat, Nevada	0.47
Texas	0.27

Average U.S. Background Radon Flux = 0.82 pCi/m^2-sec.

Source: Tr79.

Therefore, the radon released from the sub-ore surface area of the pit is 8.18×10^3 m^2 x 12 pCi of Rn/m^2-sec x 86400 sec/day = 8.48 mCi of Rn/day.

2. Radon exhalation from the overburden surface area of the pit--
    - the exposed overburden surface area (S_o) = 1.81×10^4 m^2;
    - the average radium-226 content of the overburden = 32 pCi/g; and
    - the radon flux rate for overburden is 8.7 pCi of Rn/m^2-sec.

Therefore, the radon released from the overburden surface area of the pit is 1.81×10^4 m^2 x 8.7 pCi of Rn/m^2-sec x 86400 sec/day = 13.6 mCi of Rn/day.

3.   Radon exhalation from the overburden pile remaining at the pit--

- the exposed surface area of the waste pile (S_w) = 6.33 x 10^4 m^2;

- the Ra-226 content of the surface of the overburden pile is the same as the sub-ore content = 32 pCi/g; and

- the radon flux rate for the overburden pile is 8.7 pCi of Rn/m^2-sec.

Therefore, the radon exhalation rate from the overburden pile is 6.33 x 10^4 m^2 x 8.7 pCi of Rn/m^2-sec x 86400 sec/day = 47.6 mCi of Rn/day.

The total radon release rate at the abandoned surface mine site is the sum of the above three source terms, 69.7 mCi/day. The estimated radon release rate for background soils for an undisturbed area equivalent to the surface mine area uses the following parameters:

- the ground surface area equivalent to the area of the pit opening (2.03 x 10^4 m^2) and the overburden pad area (5.98 x 10^4 m^2) = 8.01 x 10^4 m^2, and

- the radon flux rate for background soils in uranium mining areas = 0.83 pCi of Rn/m^2-sec (Table 3.72).

Therefore, the radon exhalation rate from an undisturbed area equivalent to the model surface mine is

8.01 x 10^4 m^2 x 0.83 pCi of Rn/m^2-sec x 86400 sec/day = 5.7 mCi of Rn/day.

Table 3.74 summarizes the annual radon-222 release from the model inactive uranium surface mine and all inactive uranium surface mines.

3.7.1.3 Land Surface Gamma Radiation

The surface mine uranium overlying strata must be removed in order to gain access to the uranium-bearing host materials and the ore body. The ore body consists of ore and sub-ore, and the sub-ore is simply that fraction of the ore body that contains ore uneconomical to recover. The end result of the mining is that the residues (sub-ore) enhance natural radioactive materials. That is, they are exposed or brought to the earth's surface. The enhancement will cause, in most cases, increased aboveground radiation

Table 3.74   Summary of estimated radon-222 releases from
              inactive surface mines

Source	Estimation Method	Annual Release, Ci
Mine Pit		
Sub-ore area	Model mine and limited field measurements	3.1
Overburden area	Model mine and limited field measurements	<u>5.0</u>
Total		8.1
Overburden Pile	Model mine and limited field measurements	17.4
Background	Rn-222 flux measurements and projected surface areas of model mine pit and overburden pile	2.1
Model Mine	Net Rn-222 release	23.4
All Inactive Mines	Annual net Rn-222 release from model times 1250 mines	29,000

exposure rates around the mining area. Ore and sub-ore lost through handling are subject to wind and water erosion. This effectively increases the mine site area in a radiological sense. The gamma radiation exposure levels on and around a mine site can be high enough to restrict use of the area after mining.

Gamma radiation surveys were conducted at some inactive uranium surface mining areas. Table 3.75 lists the ranges of exposure rates found. Appendix G contains more specific information concerning the surveys. The residual exposure rate levels would probably preclude unrestricted use of the pits, waste piles, and overburden.

Figure 3.25 depicts gamma radiation measurements made on radials extending outward from an inactive surface mine pit. The measurements were made with a pressurized ion chamber (PIC) at approximately 61 m intervals on each radial. As expected, the exposure rate decreases with distance away from the pit, indicating surface contamination from wind and water erosion of the spoils and ore piles. Some of the contamination may also have originated from ore and sub-ore dust losses during mining.

Since the pit resides over a former ore body and connecting or adjacent ore bodies may be located near the mine, some caution is necessary when interpreting the gamma exposure rates as indicative of surface contamination. Development drilling, indicating the presence of ore bodies, is prevalent throughout the north, west, and south areas around the pit. The northeast, east, and southeast areas around the pit have exploratory drill holes only. They indicate the probable absence of ore bodies. Although the north, northwest, west, and southwest radials cross below grade ore bodies, it is not reflected by the gamma measurements. Unless the ore body is very close to the surface, its gamma radiation will not be measured (i.e., the 1/10 value layer for earth shielding is about 0.3 m). The south radial, however, did cross an ore outcropping.

If the exposure rate measurements made at the end points of the radials (south radial excepted) are assumed to be near background, their mean value is 14.4 μR/hr with a 2 sigma error of 1.6 μ R/hr.

Assuming all measurements in excess of 14.4 + 1.6 μ R/hr or 16.0 μR/hr are a result of eroded ore and sub-ore from the mining activities, an iso-exposure rate line enclosing the eroded materials can be constructed around the mine site. The line is constructed on Fig. 3.25 and is qualitatively

Table 3.75   Summary of land surface gamma radiation surveys
in New Mexico, Texas and Wyoming

Location	Area	Gamma Radiation Exposure Rate (μ R/hr)
Poison Canyon, New Mexico	Pits	40 to 190
	Waste piles	65 to 250
	Overburden	25 to 65
Texas	Pits	5 to 400
Morton Ranch, Wyoming (1601 Pit)	Pit	16 to 63
	Ore piles	200
	Overburden	59 to 138

Source: Wo79 for New Mexico and Wyoming and Co77 for Texas.

adjusted on the south radial to compensate for the ore outcropping. The line bulges, into the southeast quadrant indicating erosion by the predominant northwest winds and contamination of about 0.3 km^2.

In summary, it appears that the residual gamma radiation exposure levels at surface mining pits and overburden piles would preclude these areas from unrestricted use. It also appears that wind and water erosions of the spoils, ore, and sub-ore are occurring and causing land contamination far removed from the mining area. Several surface mines were gamma surveyed in New Mexico. The mines could not be individually gamma radiation surveyed because of their close proximity, cross contamination from eroded ore and sub-ore, and possible ore outcrops.

GROSS GAMMA EXPDSURE RATE

Figure 3.25 Results of gamma exposure rate survey at the 1601 pit and environs, Morton Ranch uranium mine, Converse County, Wyoming (μR/hr)

3.7.2 Inactive Underground Mines

The model inactive underground mine is basically defined by dividing the total reported volumes of ore and waste removed by inactive underground mining by the number of inactive underground mines. The number of inactive underground mines has been obtained from the U.S. Department of Energy mine listing in Table 3.67. Table 3.67 lists the mines by state and type of mine. Forty-four percent of the inactive underground mines are located in Colorado, 34 percent in Utah, 9.3 percent in Arizona, and 7.0 percent in New Mexico.

For modeling purposes, we assume that there are presently 2030 inactive underground uranium mines. Table 3.69 lists the estimated underground mine waste and ore production for 1932 to 1977. Uranium mine waste and ore production statistics, on an annual basis, were available for underground producers from 1959 to 1977 (DOI59-76). Annual uranium ore production statistics for underground mining are available from 1948 to 1959 (DOE79) and from 1932 to 1942 (DOI32-42). We estimated the mine waste production for the period of 1932 to 1960 from underground mining waste-to-ore ratios and established waste-to-ore ratios using the published ore and wastes production statistics from 1959 to 1976 (DOI59-76). These ratios were fitted with a line by regression analysis in order to estimate the waste-to-ore ratios from 1932 to 1959 (Fig. 3.26). Two lines were fitted to the known waste-to-ore ratios because of the abrupt change in the ratios from 1972 to 1976. We assumed that the steeper slope was caused by increased waste production from the larger and deeper underground mines operated during this time. The estimated annual waste-to-ore ratios were multiplied by the published annual ore production values to estimate the annual waste production from 1932 to 1959. We assumed that no ore was produced from 1942 to 1948 because most of the uranium was obtained by reprocessing vanadium and radium tailings during that period (Private communication with G. C. Ritter, 1979, Bendix Field Engineering Corporation, Grand Junction, Colorado). Table 3.69 lists the cumulative annual waste production from underground mining from 1932 through 1977. The total waste produced for this period was 2.92×10^7 MT, and the total ore produced was 7.31×10^7 MT.

A simplistic way to identify a model inactive underground mine would be to divide the cumulative tonnage of ore and wastes by the number of inactive mines. We estimated the number of inactive mines from the U.S. Department of Energy mine listing (Section 2.0 and Table 3.67). The model inactive under-
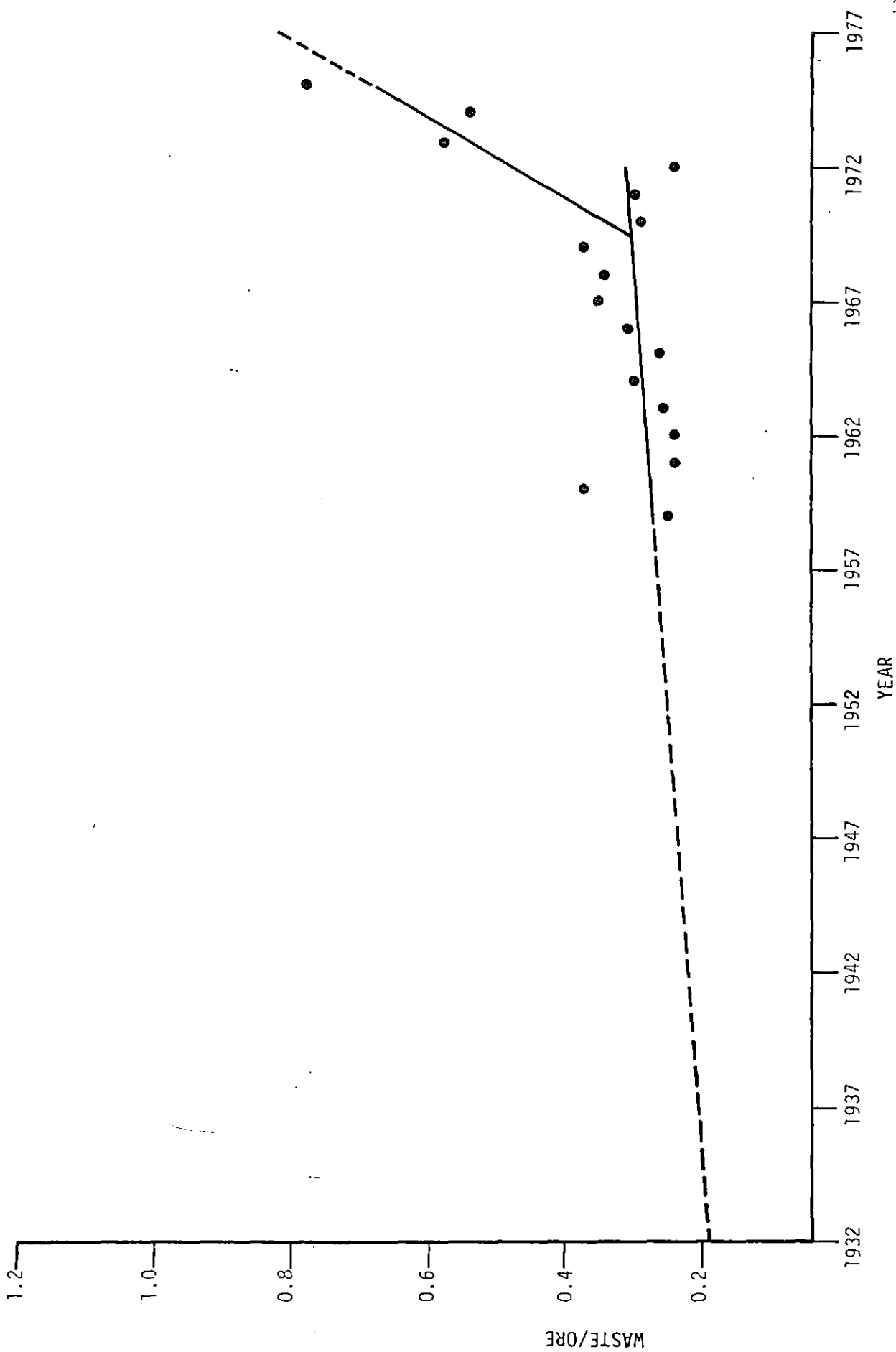
Figure 3.26 Waste to ore ratios for inactive underground uranium mines from 1932 to 1977.

ground mine produced 3.60×10^4 MT of ore and 1.44×10^4 MT of waste. Unfortunately, some of the contemporary waste and ore production has been produced by both active and inactive mines. In order to adjust the contemporary ore and waste production for that portion of the ore and wastes generated by active mining, we assumed a model active mine having a mining life of 15 years (St79). The mid-life of the mine was assumed to have occurred in 1978, with production beginning in 1971.

We also assumed that some of the mines became inactive during the 1971-1978 period and that their numbers decreased linearly. For example, 2.44×10^6 MT of ore was produced in 1972 and 85.7 percent of that ore produced was from mines that were inactive by 1978. Therefore, adjusted ore production was 2.09×10^6 MT for 1971. The ore production for 1973 was 1.15×10^6 MT and 1.27×10^6 MT in 1974; 1.06×10^6 MT in 1975; 1.02×10^6 MT in 1976; and 6.16×10^5 MT in 1977. The adjusted waste production was: 5.08×10^5 MT in 1972; 6.67×10^5 MT in 1973; 8.13×10^5 MT in 1974; 9.43×10^5 MT in 1975; 7.43×10^5 MT in 1976; and 4.99×10^5 MT in 1977.

Through 1978, the cumulative adjusted ore production from inactive underground mines was 6.37×10^7 MT, and the cumulative adjusted waste production was 2.04×10^7 MT. The model inactive underground mine was assumed to have produced 3.14×10^4 MT of ore and 1.00×10^4 MT of waste. Assuming a density of 2.0 MT per m^3, the volume of ore and waste removed were 1.6×10^4 and 5.0×10^3 m^3, respectively.

Fifty percent of the waste volume mined we assumed to be sub-ore. The volume of waste rock (i.e., containing no sub-ore) removed during the mining is 2.5×10^3 m^3. Assuming an entry dimension of 1.83 m x 2.13 m, about 615 m of shafts and haulways are in the model mine. The ore body we assumed to have an average thickness of 1.8 m with a length and width of 91.2 m each. The surface area of the passages would be 4.83×10^3 m^2. The surface area of the mined-out ore body would be 1.71×10^4 m^2.

3.7.2.1 Waste Rock Piles

Wastes produced from underground uranium mining were generally cast or dumped near the mine entries. Those wastes that were dumped on relatively flat terrain formed dome-shaped piles. Wastes cast from rim mines generally formed long, thin sheets down the canyon slopes. Since most of the inactive underground mines are in the Uravan Mineral Belt, the waste pile shape (dome)

is assumed to be predominant (see Appendix G.1.2) and is used for the calculations of the waste pile dimensions.

The waste produced at a typical underground mine consists of waste rock and sub-ore. The waste rock is assumed to be on the bottom of the waste pile since it was generally removed first. Sub-ore, which was removed later, is assumed to cover or plate the waste pile. The waste piles are assumed to be dome shaped, covering a circular area of 0.40 hectares. The dome is assumed to be a spherical segment with a height (b) and base (c) of 71.8 m. The volume (V) of the spherical segment, 6.3×10^3 m^3 when corrected for bulking, is equal to the volume of wastes and is expressed as

$$V = \frac{1}{24} \pi b (3c^2 + 4b^2). \tag{3.30}$$

The surface area of the spherical segment is given by the expression

$$S = \frac{1}{4} \pi (4b^2 + c^2) \text{ where } S = \text{Surface area } (m^2). \tag{3.31}$$

The term b is solved by substitution and iteration in the former equation and is substituted in the latter equation to determine the surface area of the wastes:

$$V = 6.3 \times 10^3 \ m^3 = \frac{1}{24} \pi b (15465 + 4b^2) \text{ where: } b = 3.1 \text{ m.} \tag{3.32}$$

The surface area of the waste pile is

$$S = \frac{1}{4} \pi (4b^2 + c^2) \tag{3.33}$$

$$= \frac{1}{4} (3.14) (38 + 5155)$$

$$= 4.08 \times 10^3 \ m^2.$$

The thickness (T) of sub-ore on the surface of the waste pile is

$$\frac{\text{volume of sub-ore}}{\text{area of waste pile}} = \frac{3.2 \times 10^3 \ m^3}{4.08 \times 10^3 m^2} = 0.78 \text{ m.} \tag{3.34}$$

In summary, the waste pile at an inactive underground uranium mine is assumed to have the shape of a spherical segment with a surface area 4.08 x 10^3 m^2. The pile is assumed to have an inner core of waste rock covered or plated with 0.78 m of sub-ore on its exposed surface. It is expected that the plate of sub-ore on the waste pile would be more pronounced than the sub-ore plates on overburden piles at surface mines because of diminished

blending, mining practices, and the lower waste-to-ore ratio. The grand average of the radium-226 concentrations in the waste rock and overburden piles (Table 3.72) appear to confirm this expectation.

Table 3.76 lists average annual emissions of contaminants due to wind erosion of the waste rock pile. These values were estimated by multiplying an emission factor of 2.12 MT/hectare-yr, derived in Appendix I, by the waste pile surface area, 0.408 hectares, and the stable element concentrations given in Table 3.19. We assumed uranium and thorium concentrations to be 110 pCi/g and 2 pCi/g, respectively.

Table 3.76 Average annual emissions of radionuclides (μCi) and stable elements (kg) in wind suspended dust at the model inactive underground mine

Contaminant	Waste Rock Pile[a]	Contaminant	Waste Rock Pile[a]
Arsenic	0.07	Molybdenum	0.10
Barium	0.80	Nickel	0.02
Cadmium	ND[b]	Lead	0.07
Cobalt	0.01	Ruthenium	ND
Copper	0.05	Selenium	0.10
Chromium	0.02	Strontium	0.11
Iron	14	Vanadium	1.2
Mercury	ND	Zinc	0.03
Potassium	22	Uranium-238 and each	
Magnesium	3.0	daughter	238
Manganese	0.83	Thorium-232 and each	
		daughter	1.7

[a]Mass emissions = 8.65 x 10^5 g/yr.

[b]ND - Not detected.

3.7.2.2   Radon-222 from the Mine Area

We estimated the total radon released from the model inactive underground mine from the following parameters:

1.   Radon exhalation from the waste rock pile--

    • the exposed surface area of the waste pile = 4.1×10^3 m^2;

    • the average Ra-226 content of the waste pile is 110 pCi/g; and

    • the radon flux rate for the waste pile is 13 pCi of Rn/m^2-sec.

Therefore, the radon released from the waste pile is

$$4.1 \times 10^3 \; m^2 \times 13 \; pCi \; of \; Rn/m^2\text{-sec} \times 86400 \; sec/day = 4.6 \; mCi \; of \; Rn/day.$$

2.   Typical background release rate--

    • the ground surface area equivalent to the area covered by the waste pile = 4.1×10^3 m^2, and

    • the radon flux rate for background soils in uranium mining areas = 0.83 pCi of Rn/m^2-sec (Table 3.72).

Therefore, the radon exhalation rate from an undisturbed area equivalent to the waste pile of a model underground mine is

$$4.1 \times 10^3 \; m^2 \times 0.83 \; pCi \; of \; Rn/m^2\text{-sec} \times 86400 \; sec/day = 0.29 \; mCi \; of \; Rn/day.$$

The net radon release rate due to the waste pile at the inactive underground mine is 4.6 minus 0.29 or about 4.3 mCi of Rn/day above normal background.

Natural ventilation will occur in most mines and usually is considered by mine ventilation engineers when planning the forced ventilation systems. The natural force that can maintain a natural air flow due to temperature differences is thermal energy. The thermal energy added to a system is converted into a pressure difference. If the pressure difference is sufficient to overcome head losses, a flow of air will occur.

Natural ventilation depends upon the difference between the temperature inside and outside of a mine and the difference between the elevation of the mine workings and the surface. Air flow by natural ventilation is generally small (140 - 566 m^3/min) in shallow mines (Pe52). In deep mines, natural ventilation flows may range from 1,420 to 4,250 m^3/min (Pe52). The flow in either the shallow or deep mines depends upon the depth, size, and number of

openings. The intensity of thermal energy-induced natural pressure usually ranges from a few hundredths to a few tenths cm of water in shallow (less than 460 m deep) mines (Pe52). The maximum pressure drop per 305 m of depth in deep mines is about 2.54 cm of water in winter and about 0.84 cm during the summer (Pe52).

In general, natural ventilation is subject to considerable fluctuation. It usually increases to a maximum in winter and a minimum in summer for deep mines. The typical inactive underground uranium mine would be shallow; therefore, the natural ventilation would be expected to reach its maximum in the winter and summer and its minimum in the spring and fall (air temperature in the mine closely approaches the outside temperature during the spring and fall).

A first approximation of the annual release of Rn-222 from an inactive underground mine simply would be that all Rn-222 released into the mine air will be exhausted by natural ventilation before a significant radioactive decay occurs. That is, the quantity of radon released into the mine is equal to the quantity of radon released from the mine. The quantity of Rn-222 released from the sub-ore surfaces remaining in the mined-out ore body is

$$Q \frac{(pCi\ Rn\text{-}222)}{sec} = A \times \phi_{so}, \tag{3.35}$$

where A is the surface area of the mined out ore body (m^2)

ϕ_{so} = exhalation rate of the Rn-222 from sub-ore per unit area per unit time, 12 $\frac{pCi}{m^2\text{-}sec}$ (Section 3.7.1.2)

$$Q = (1.71 \times 10^4\ m^2) \frac{(12\ pCi)}{m^2\text{-}sec} = 2.1 \times 10^5\ pCi/sec.$$

It should be noted that ϕ_{so} is the average radon flux physically measured from sub-ore bodies in inactive surface mines (Section 3.7.1.2). Because of safety considerations, no measurements were made from sub-ore bodies in inactive underground mines during the April 1979 field surveys. The annual Rn-222 source term from the mined-out ore body in an inactive underground uranium mine, using the preceding assumptions, is

$$Q \frac{(Ci)}{yr} = 2.1 \times 10^5 \frac{pCi}{sec} \times 3.6 \times 10^3 \frac{sec}{hr} \times 24 \frac{hr}{d} \times 365 \frac{d}{yr} \times$$

$$\frac{1}{10^{12} \frac{pCi}{Ci}} = 6.6 \frac{Ci}{yr} . \tag{3.36}$$

The annual Rn-222 source term (Q) from the passageways, assuming an exhalation rate of $8.7 \frac{pCi}{m^2\text{-}sec}$ for overburden, is (Section 3.7.1.2)

$$(4.8 \times 10^3 \text{ m}^2) (2.7 \times 10^{-4} \frac{Ci}{yr\text{-}m^2}) = 1.3 \frac{Ci}{yr} \tag{3.37}$$

The air flow rate from the mine, assuming 140 m^3/min for an average shallow mine, will exchange the mine air every three hours. The average annual radon-222 concentration will be

$$7.9 \text{ Ci/yr} \times \frac{1}{7.4 \times 10^7 \text{ m}^3/yr} \times \frac{1}{1000 \text{ } \ell/m^3} = 107 \text{ pCi/}\ell.$$

The radon daughter concentration will be about 87 percent of equilibrium with the radon, assuming a mean residence time of the radon in the mine to be 1.5 hours.

Several inactive mines in the Grants, New Mexico area were monitored for radon discharges by natural ventilation. One of the mines monitored was relatively small and had a vertical shaft access. Five cased 30 cm diameter vents were found and were assumed to be connected with the mine. The shaft was covered with steel plate, but access holes were cut in the plate and one corner had been pried up. Four vents were capped with buckets. Just one cover was gas tight. One vent was partially covered with a piece of wood. Only very small flow rates due to natural ventilation were measured at the shaft and vents. The maximum radon emission from the mine per day was estimated to be 2.8×10^3 µCi. This low radon discharge rate is probably due to partial blockage of the vents and water in the mine. The mine was partially flooded, and flowing water could be seen at the bottom of the shaft. The

effect of the water would be to partially or completely close off the mine workings and substantially reduce natural ventilation. The water would also dissolve and substantially suppress the radon exhaling from the surface areas of the mine. Thus, we believe that the radon discharges from wet inactive mines via natural ventilation will be minimal.

Investigation at another inactive mine revealed that it was connected to three other inactive mines that were subsequently connected to two active mines. Ventilation fans at the connecting active mines were usually shut down after the end of the day shift and on weekends. Mine air was exhausted by natural ventilation through the shaft (highest opening) and vents of the mine investigated. A flow rate up to 88 m^3/min was observed coming from the shaft, and radon-222 concentrations reached 11,000 pCi/ℓ. The average flow rate observed over a weekend was 75 m^3/min, with an average radon-222 concentration of 9,800 pCi/ℓ. The average radon emission was 1.1 Ci/day.

Figure 3.27 is a plot of the changes in the Rn-222 concentration in the air from the shaft of the inactive mine investigated. The average of the measurements of the air flow rate from the shaft was about 76 m^3/min. The Rn-222 concentration increased almost linearly with time for about 20 hours after the fans were shut down at the end of the day shift on April 27, 1979. The Rn-222 concentrations also leveled off at about 10,000 pCi/ℓ. A dip, presumed to have been caused by high winds, occurred in the Rn-222 concentration curve from about 1000 to 1600 hours on April 28, 1979.

Since the curve is relatively flat at 10,000 pCi/ℓ, it is assumed that the rate of production of the Rn-222 is equal to the rate of removal of the Rn-222 from the six mines. The average residence time of the radon in the mine air is assumed to be approximately 10 hours, and the radon daughters would be in near-equilibrium (assumed to be ≅ 90 percent). Assuming that all six interconnecting mines contributed equally to the source term measured, the release rate of Rn-222 for a single mine will be

10,000 pCi/ℓ x 76,000 ℓ/min x 1440 min/day x 10^{-12}Ci/pCi ÷ 6

= 0.18 Ci/day.

Based on the preceding estimation of Rn-222 and progeny released from a typical mine on the Colorado plateau and physical measurements at six connected mines, the annual radon release rate may range from 7.9 to 66 Ci/yr. These source term estimates, of course, are based on a single mine. Many mine workings are, in fact, interconnected. If these interconnected workings are assumed to constitute a single mine, then the upper limit of Rn-222 and
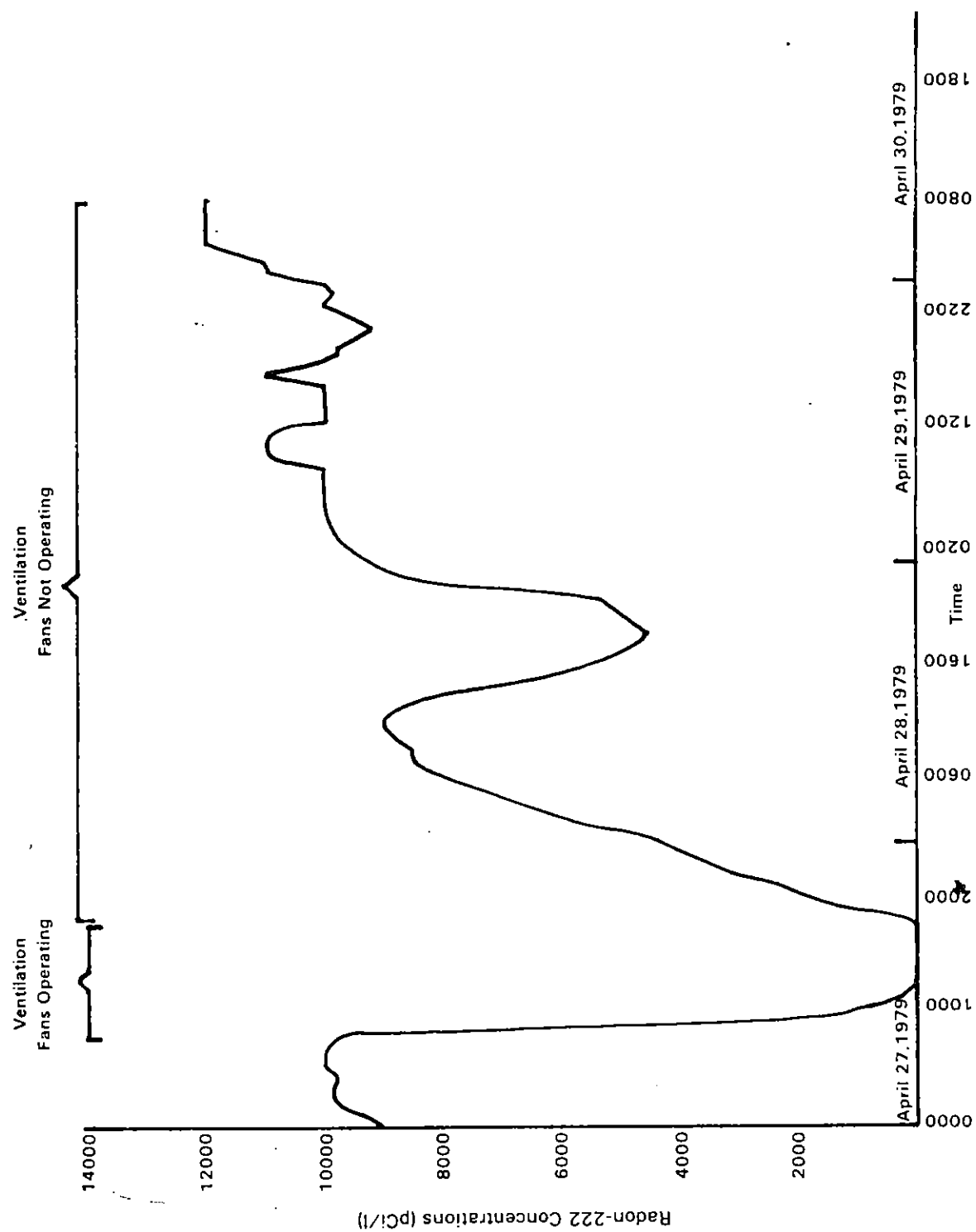
Figure 3.27 Radon-222 concentrations in mine air discharged by natural ventilation.

progeny discharge known at this time will be about 10,000 pCi/ℓ with an annual Rn-222 source term of about 400 Ci/yr. For example, 67 percent of all inactive underground uranium mines are in or near the Uravan mineral belt and are probably dry. Their aggregate Rn-222 discharge by natural ventilation is estimated to be

$$1360 \text{ mines} \times 66 \text{ Ci } \frac{Rn\text{-}222}{yr\text{-}mine} = 9.0 \times 10^4 \text{ Ci/yr.}$$

In summary, there is little information available on the discharge of Rn-222 and its progeny from the vents and entries of inactive uranium mines . by natural ventilation. Some physical measurements indicate that the discharges may be substantial. It is known, through surveys conducted to support this study, that a large majority of the inactive uranium mines are not isolated from the atmosphere and are capable of discharging their Rn-222 and progeny into the local environment. It is also known that some self-sealing will probably occur at some of the mines, due to flooding, cave-ins, and subsidence. Table 3.77 summarizes estimates of the annual radon-222 releases from inactive underground uranium mines. This potential source of exposure could be practically eliminated by proper sealing of the inactive mines.

3.7.2.3 Land Surface Gamma Radiation

Gamma radiation surveys were conducted around underground mining areas in Colorado and New Mexico. Table 3.78 lists the ranges of gamma radiation exposure rates measured at some of the mines. The elevated gamma ray exposure rates on the waste piles are due primarily to plating those piles with sub-ore removed during the mining process.

Some radioactive materials originating from ore and sub-ore handling can be lost into the local environment around a mine site. Erosion of the mine wastes can also disperse contaminants into the local environment. Figure 3.28 illustrates gross gamma radiation exposure rate measurements around an inactive underground uranium mine in New Mexico. Background gamma-ray exposure rate measurements made around the mine area ranged from 12 to 15 μR/hr. According to the measurements made, exposure rate levels exceeded background from 50 to more than 100 meters from the waste piles. The area that has been contaminated far exceeds the area physically disturbed at the mine site. Gross gamma exposure rates measured on the waste piles averaged about 95

Table 3.77   Summary of radon-222 releases from inactive underground mines

Source	Estimation Methods	Annual Release, Ci
Model Mine		
Waste Rock Piles	Calculated volume & surface area; limited field measurements of radon flux	1.7
Underground workings	Radon release based on natural ventilation rate for shallow mines	
Sub-ore Surfaces	Calculated surface area; limited radon flux measurements of sub-ore	6.6
Passageways	Calculated passageway surface area; limited measurements of radon flux from overburden	1.3
Background	Field measurements of radon flux; and projected area of waste rock pile	0.11
Model Mine	Total radon source minus background	9.5
Actual Mine		
Underground workings (dry)	Field measurements	66
Underground workings (wet)	Field measurements	1.1
Waste rock piles	Calculated volume & surface area; Limited field measurements of radon flux	1.7

Table 3.78   Summary of land surface gamma radiation surveys in
                    Colorado and New Mexico

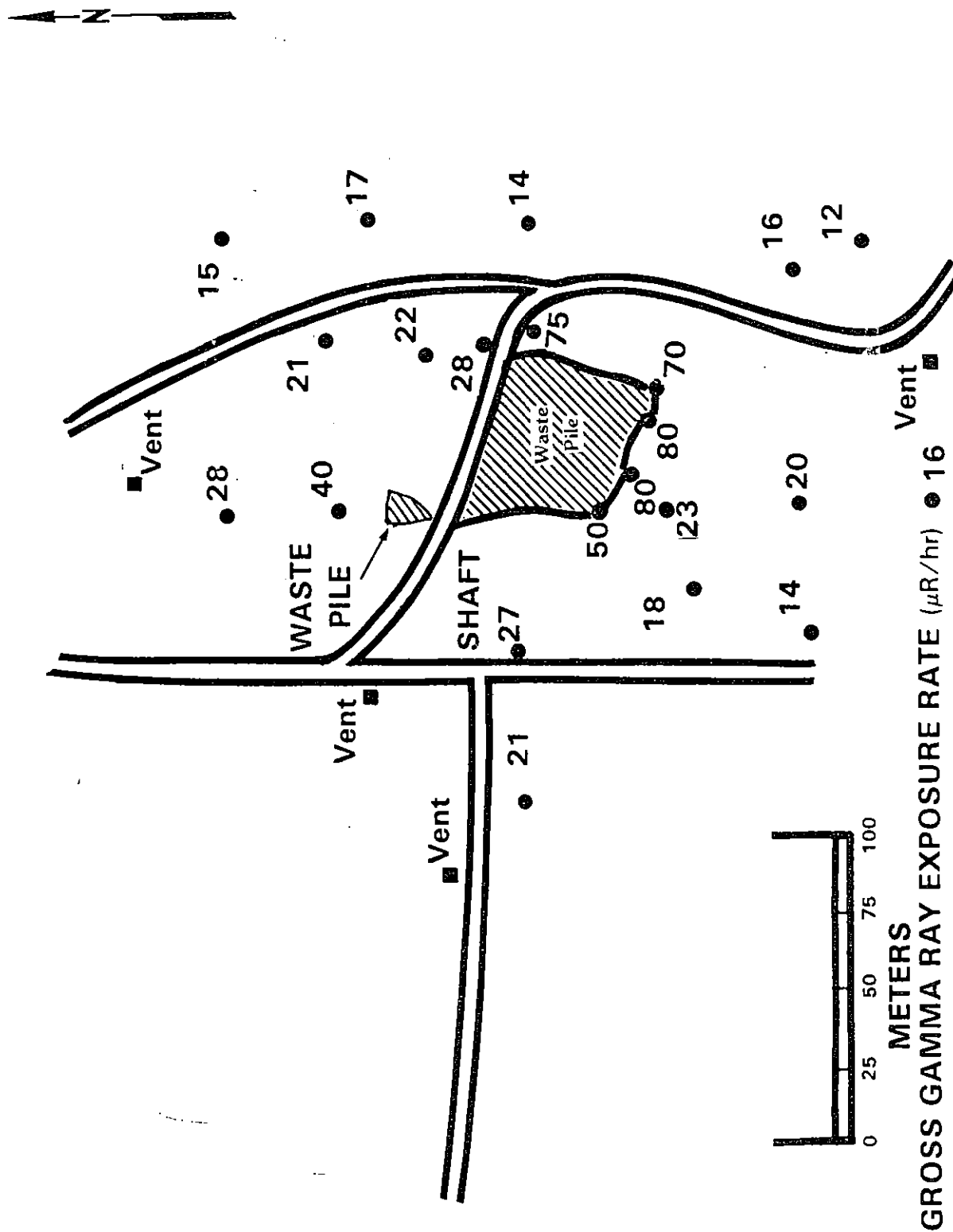| Location	Area	Gamma Radiation Exposure Rate (μ R/hr)
Boulder, Colorado	Waste piles	40 to 100
Uravan, Colorado	Waste piles	50 to 220
San Mateo, New Mexico	Waste pile	35 to 275
	Ore	100 to 350
	Overburden	20 to 120
	Background	10 to 13
Mesa Top Mines, New Mexico	Waste piles	25 to 290
Barbara J #1 Mine, New Mexico	Waste piles	21 to 170
	Background	12 to 15

Source: Wo79.

Figure 3.28 Gamma radiation survey around an inactive underground uranium mine in New Mexico.

GROSS GAMMA RAY EXPOSURE RATE (μR/hr)

METERS

µR/hr, which would make them unsuitable for unrestricted use.

In summary, wastes from underground uranium mining technologically enhance natural radioactivity and may .be considered low-level radioactive wastes. Improperly controlled wastes will be dispersed into the surrounding environment by the mining activities and erosion.

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